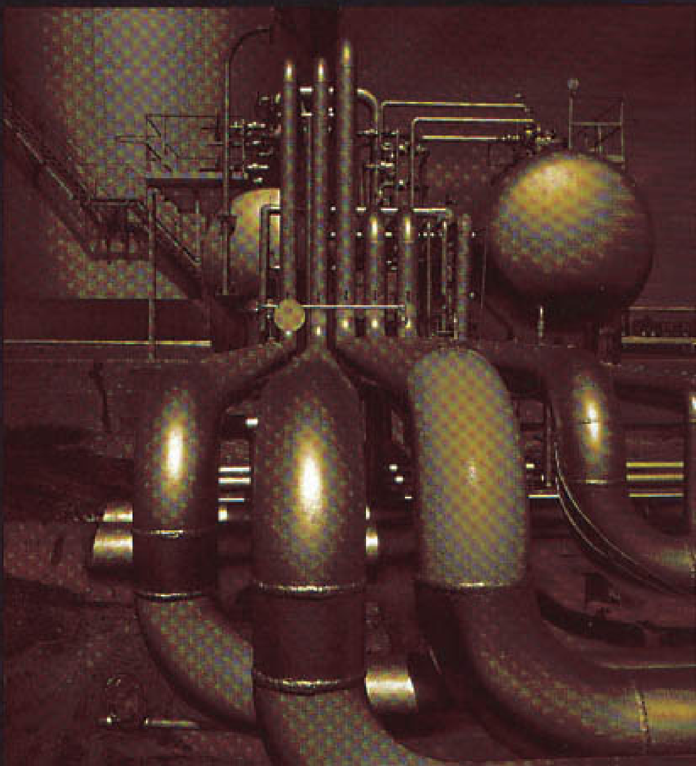
A Handbook of Industrial Ecology

Edited by
Robert U. Ayres
and
Leslie W. Ayres



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Preface

It is customary for a volume like this to start with a preface. I have not had a course in preface writing ('preface-ology 101'), but I suppose the preface must be analogous to materials given out on freshman orientation day, where a new college student learns where the most important college institutions are to be found, such as the student union, the gym, the football stadium, the dormitories, the dining hall and – incidentally – the library, the bookstore, the lecture halls and the chemistry lab (where that hydrogen sulfide smell seems to be coming from).

The confusion is compounded by the fact that there are two of us, but only one (RUA) is writing this. Thus the pronouns will be seen to wander erratically.

I have put it off until the very last moment in hopes of some inspiration. But the sad fact is, nobody ever quotes from, or remembers what is written in, a preface. I suspect that nobody ever reads prefaces. I don't. Why, then, should I write one?

Thinking out loud (so to speak), is the preface needed to define the subject? We have assigned the opening chapter of this volume to two authors who clearly believe in formal definitions and statements of purpose, and who have been instrumental (with others) in creating a formal graduate program in industrial ecology, and a professional journal in the subject. Apparently IE is now a subject. Our own view of what belongs within the boundaries of IE is implicit in the structure of this volume. Readers will note that this tends to lean towards inclusion rather than exclusion. Enough said.

Should the preface be used to provide the historical background to the subject of the volume? Again I think not. There are two chapters (numbers 2 and 3) explicitly included for this purpose. They do the job better than we could. Shall I use the preface as an opportunity to pontificate on the future of our subject? The trouble is I can't believe the future cares what I think. (For that matter, I don't care much about what the future thinks.) So, I have no inspiration along those lines.

Is the preface needed to explain the origin of the volume itself? At one level, that is easily covered in nine words: the publisher asked us to do it. We agreed. Why did we agree? I'm still asking myself that one. Perhaps it was for the big money. (I don't want to embarrass him by calculating exactly what that worked out to per hour.) Well, there's also our leather-bound copy inscribed on vellum and signed by Edward Elgar personally in gold ink. He hasn't sent that yet, but we're saving a space for it.

Seriously, I suppose the main reason we took it on was because I think the subject is finally coming of age. If it is time for a journal and a professional society (in progress), then it is also time for a *Handbook* (perhaps, soon, even an *Encyclopedia*) to bring together the leading thinkers and practitioners in the newly emerging field and give them a chance to present 'the state of the art' between two covers. In short, somebody had to do it and I've been around the subject longer than most. The other reason was that I wanted to give myself an excuse to read and understand all the stuff I've been accumulating on my shelves for a decade or more, especially a number of relevant PhD theses that have appeared in recent years. Summaries of several of them appear in the following chapters.

Having done it, have I learned anything worth passing on to the next generation of editors? I don't really know what is worthy of passing on, but a couple of subversive thoughts have struck me about what I would try to do differently next time, if there were a next time (perish the thought).

The first of my subversive thoughts is that the subject of data gathering and data processing is routinely under-represented. By the same token, 'modeling' is equally over-represented. For some reason young PhD students are brought into the world under the impression that their task is to massage a pile of putty-like data into a model, which can then be baked in an oven (so to speak) and take its place with hundreds of other oven-baked models stored in a warehouse (library), where they will be checked out from time to time and marveled over by model connoisseurs. Editorial observation: in reality, most soft models are used but once, usually to secure a PhD degree, and never seen or heard from again. On the other hand, lowly data are likely to be recycled many times, often by people who have forgotten or never knew the source. The great danger, in this, is to confuse 'raw' data with 'model' data.

Of course, in established fields, like electrical engineering or chemical engineering, much effort has gone into distinguishing the two categories. Raw data are obtained by direct measurement. Processed data are picked over to eliminate outliers and averaged or otherwise modified. Model data are calculated from processed data. There are handbooks consisting almost solely of data series, revised and updated at regular intervals by committees of experts.

IE lacks any such tradition. There is no clear distinction between 'hard' data and 'soft' data. In recent years there has been an outbreak of national mass-flow studies, a number of which are represented in this volume. Missing from those studies, in general, is any discussion of the sources of the data, or the reliability of those sources. Particularly absent is any distinction between what is calculated (and how), what is measured (and how) and what other choices might have been made. Hardly any reader would realize that mass-flow data are rarely measured (or measurable) directly, and that each beautiful flow chart is, in fact, a model. No problem, except that the models used are presented as data. Detailed discussions of sources and assumptions are mostly absent.

As a rough generalization, one end of each mass flow is calculated or estimated based on some model, whether explicit or implicit. If the inputs to an industrial process are well known from measurements, the outputs (especially emissions) are generally estimated by means of some combination of mass balance and process analysis. If the outputs of a process are counted (or surveyed) accurately – for instance, food consumption – it is still very uncertain how the outputs are derived from raw material inputs. In principle, every mass flow should be verified both 'top down' (from aggregate data) and 'bottom up' from process–product data. In practice, we are a long way from this ideal situation. In short, as the field of IE matures, it will be necessary to do something about classifying and improving the underlying data bases.

Another observation: there is a remarkable tendency on the part of many authors to justify their work in terms of 'policy needs'. I do not doubt that policy makers at all levels need better models (and better data). However, the majority of the models discussed in this volume are far from being either widely enough accepted, or user-friendly enough, to be immediately useful to policy makers now. This is not a criticism. It is where the field stands. What is missing, and badly needed in my opinion, is a greater focus on the gaps. What data are missing but needed? How might they be obtained? What models are in need of verification? How might it be done? What models are in need of radical improvement? How might it be approached? What models are likely to be misleading or misused? How can that be avoided?

A few other remarks of a more mundane nature come to mind. It is customary for an editor of a large multi-author volume such as this to appoint up to half-a-dozen associate editors. This group then either reviews the draft articles themselves, or farms them out for review, usually to other authors. Since the group is largely self-selected and mutually acquainted through years of meetings and conferences, authors can often guess who is writing the anonymous review. In consequence, reviews by colleagues tend to be quite bland and uncritical. Since cross-reviewing is an unpaid chore, it is often at the end of a long queue. Then the reviews are collected by the editor, who adds a few words of advice (but the article is already accepted, by definition) and are sent back to the authors.

Books edited in this way seldom appear in print in much less than three years after the original idea of the book is broached or accepted – as the case may be – by the publisher. The long lag time is, in itself, one of the disincentives to promptness on the part of experienced authors, who know that, however late they are, they will not be the last. Usually, they are right, since only one author can be the last and there is no booby prize.

In this *Handbook* we have tried hard to break the pattern. The *Handbook* idea was proposed by Edward Elgar about a year ago. We solicited our authors, suggesting specific topics, in March 2000. All but a few accepted quite promptly. A June deadline was proposed, on the argument that, if you are going to do it, do it, sooner is better than later. We expected some delays, of course. Only about half of the authors actually met the deadline, but in July we were able to start on the hard part, which was to read each of the drafts critically. In almost every case, significant cuts were requested (and, in a few cases, small additions.)

For the non-English speaking authors, we soon found myself doing what newspaper and magazine editors routinely do, which is to rewrite. Not to change the authors' intention, but to express it more clearly and more efficiently, in (many) fewer words. In a few cases the rewrite was fairly drastic. Of course, the rewritten articles were sent back to authors for approval or further changes. Most were polite about it, and some even thanked us for our efforts (for which we duly express our gratitude here and now). If there is any author who feels that we trampled over his/her 'rights' of free expression, but who kept quiet about it, we can only apologize for hurt feelings and point out that the space limitation was necessary and we didn't enjoy doing the extra work either. I do believe the result is more readable, however.

One other innovation, also to save space, was to move all the references to the end of the volume in a single composite list. This cost quite a bit of work, but makes a better result.

This is probably the place to mention my personal regret that several of the pioneers of industrial ecology are not adequately represented in this volume, except insofar as they appear in the citations at the end. I will not mention specific absentees for fear of offending others. A few of you were invited and begged off. (Probably you are already famous enough.) A few others I missed for various good reasons, such as lack of an address, or bad reasons such as plain and simple oversight. So, here's another apology to absent friends.

ROBERT U. AYRES

PART I

Context and History

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1. Industrial ecology: goals and definitions

Reid Lifset and Thomas E. Graedel

Setting out the goals and boundaries of an emerging field is a hapless task. Set them too conservatively and the potential of the field is thwarted. Set them too expansively and the field loses its distinctive identity. Spend too much time on this task and scarce resources may be diverted from making concrete progress in the field.

But in a field with a name as provocative and oxymoronic as industrial ecology, the description of the goals and definitions is crucial. Hence this introductory chapter describes the field of industrial ecology, identifying its key topics, characteristic approaches and tools. The objective is to provide a map of the endeavors that comprise industrial ecology and how those endeavors relate to each other. In doing so, we seek to provide a common basis of discussion, allowing us then to delve into more conceptual discussions of the nature of the field.

No field has unanimity on goals and boundaries. A field as new and as ambitious as industrial ecology surely has a long way to go to achieve even a measure of consensus on these matters, but, as we hope this chapter shows, there is much that is coalescing in research, analysis and practice.

DEFINING INDUSTRIAL ECOLOGY

The very name *industrial ecology* conveys some of the content of the field. Industrial ecology is *industrial* in that it focuses on product design and manufacturing processes. It views firms as agents for environmental improvement because they possess the technological expertise that is critical to the successful execution of environmentally informed design of products and processes. Industry, as the portion of society that produces most goods and services, is a focus because it is an important but not exclusive source of environmental damage.

Industrial ecology is *ecological* in at least two senses. As argued in the seminal publication by Frosch and Gallopoulos (1989) that did much to coalesce this field, industrial ecology looks to non-human 'natural' ecosystems as models for industrial activity.[1] This is what some researchers have dubbed the 'biological analogy' (Wernick and Ausubel 1997; Allenby and Cooper 1994). Many biological ecosystems are especially effective at recycling resources and thus are held out as exemplars for efficient cycling of materials and energy in industry. The most conspicuous example of industrial re-use and recycling is an increasingly famous industrial district in Kalundborg, Denmark (Ehrenfeld and Gertler 1997; Chapter 27). The district contains a cluster of industrial facilities including an oil refinery, a power plant, a pharmaceutical fermentation plant and a wallboard factory. These facilities exchange by-products and what would otherwise be called wastes. The network of exchanges has been dubbed 'industrial symbiosis' as an explicit analogy to the mutually beneficial relationships found in nature and labeled as symbiotic by biologists.

Second, industrial ecology places human technological activity – industry in the widest sense – in the context of the larger ecosystems that support it, examining the sources of resources used in society and the sinks that may act to absorb or detoxify wastes. This latter sense of 'ecological' links industrial ecology to questions of carrying capacity and ecological resilience, asking whether, how and to what degree technological society is perturbing or undermining the ecosystems that provide critical services to humanity. Put more simply, economic systems are viewed, not in isolation from their surrounding systems, but in concert with them.

Robert White, the former president of the US National Academy of Engineering, summarized these elements by defining industrial ecology as . . . 'the study of the flows of materials and energy in industrial and consumer activities, of the effects of these flows on the environment, and of the influences of economic, political, regulatory, and social factors on the flow, use, and transformation of resources' (White 1994).

This broad description of the content of industrial ecology can be made more concrete by examining core elements or foci in the field:

- the biological analogy,
- the use of systems perspectives,
- the role of technological change,
- the role of companies,
- dematerialization and eco-efficiency, and
- forward-looking research and practice.

The Biological Analogy

The biological analogy has been applied principally at the level of facilities, districts and regions, using notions borrowed from ecosystem ecology regarding the flow and especially the cycling of materials, nutrients and energy in ecosystems as a potential model for relationships between facilities and firms. The archetypal example is the industrial symbiosis in Kalundborg, but the search for other such arrangements and even more conspicuously the effort to establish such symbiotic networks is emblematic of industrial ecology – so much so that many with only passing familiarity of the field have mistakenly thought that industrial ecology focused only on efforts to establish eco-industrial parks.

This analogy has been posited more generically as well, not merely with respect to geographically adjacent facilities. Graedel and Allenby (1995) have offered a typology of ecosystems varying according to the degree to which they rely on external inputs (energy and materials) and on release of wastes to an external environment. Expressed another way, the ecosystems vary according to the linearity of their resource flows as shown in Figure 1.1: type I is the most linear and reliant on external resources and sinks; type III stands at the other extreme, having the greatest degree of cycling and least reliance on external resources and sinks. The efficient cycling of resources in a biological system is held out as an ideal for industrial systems at many scales. This framework thus connects the biological analogy to strong emphasis in industrial ecology on the importance of closing materials cycles or 'loop closing'.

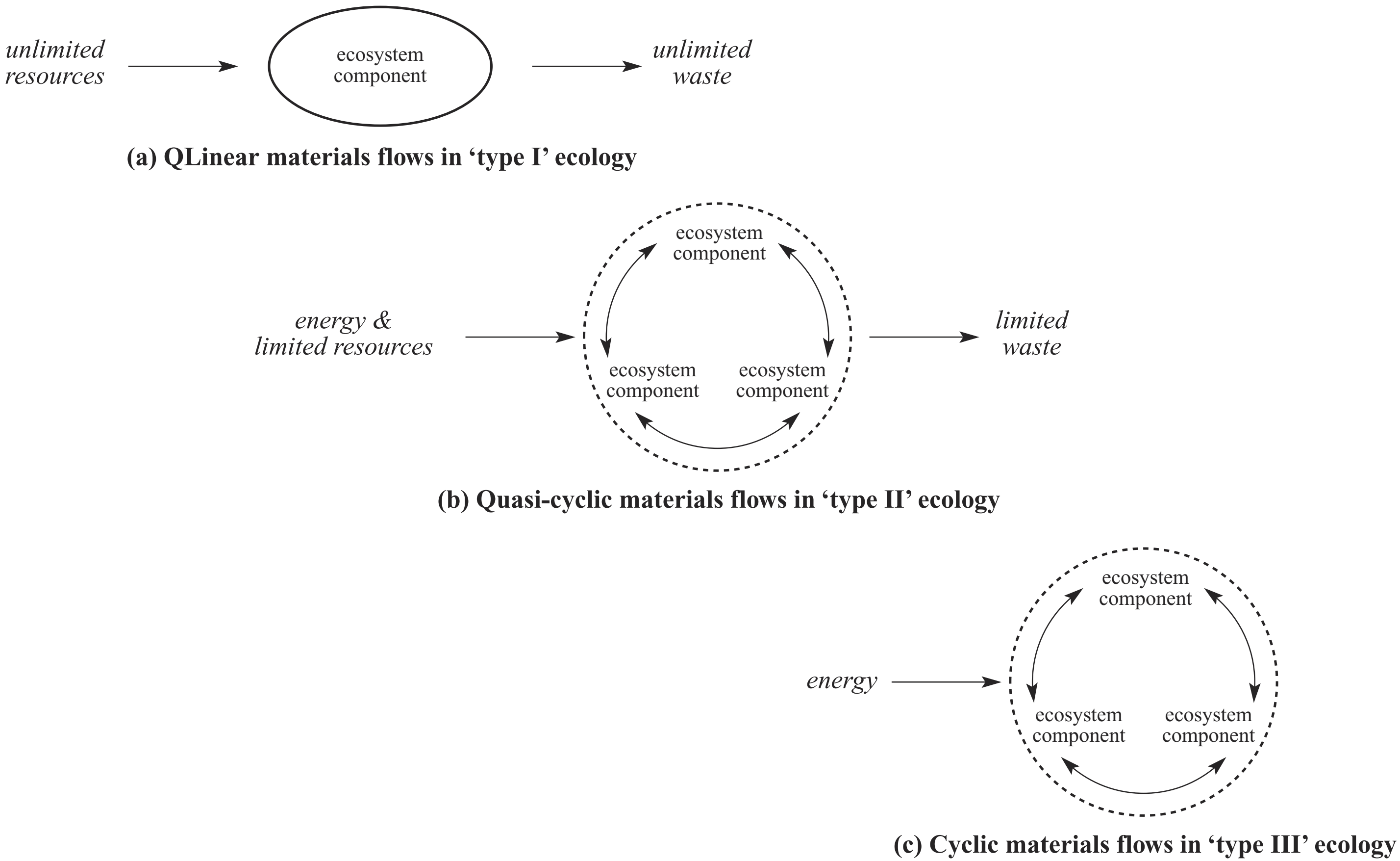

Figure 1.1 Typology of ecosystems

The biological analogy has been explored in other ways. The ecological analogy has, for example, been applied to products as a source of design inspiration (Benyus 1997), as a framework for characterizing product relationships (Levine 1999) and as a model for organizational interactions in technological 'food webs' at the sector or regional levels (Graedel 1996; Frosch *et al.* 1997).

The analogy to ecology is suggestive in other respects (Ehrenfeld 1997). It points to the concepts of community and diversity and its contribution to system resilience and stability as fundamental properties of ecosystems – and as possible models of a different sort for industrial activity. These dimensions of the analogy may point to ways to integrate organizational aspects of environmental management more deeply into the core of industrial ecology, but they have not been as extensively explored as the use of ecosystems ecology with its emphasis on flows and cycling of resources. As Andrews (2000) points out, there are long-standing bodies of scholarship that apply the ecological notions directly to social, as opposed to technological, dimensions of human activity including organizational, human and political ecology. The biological analogy is not confined to ecological similes. A more quantitative embodiment of the biological analogy is the metabolic metaphor that informs materials flow analysis (see below) by analogizing firms, regions, industries or economies with the metabolism of an organism (Ayres and Simonis 1994; Fischer-Kowalski 1998; Fischer-Kowalski and Hüttler 1998). Whether or not there is a significant difference between the ecological and metabolic metaphors is a matter of friendly dispute. For one view, see Erkman (1997).

Systems Perspective

Industrial ecology emphasizes the critical need for a systems perspective in environmental analysis and decision making. The goal is to avoid narrow, partial analyses that can overlook important variables and, more importantly, lead to unintended consequences. The systems orientation is manifested in several different forms:

- use of a life cycle perspective,
- use of materials and energy flow analysis,
- use of systems modeling, and
- sympathy for multidisciplinary and interdisciplinary research and analysis.

The effort to use a life cycle perspective, that is, to examine the environmental impacts of products, processes, facilities or services from resource extraction through manufacture to consumption and finally to waste management, is reflected both in the use of formal methods such as life cycle assessment (LCA) and in attention to approaches that imply this cradle-to-grave perspective and apply it in managerial and policy settings as well as in research contexts. This latter group includes product chain analysis (Wisberg and Clift 1999), integrated product policy (IPP, also known as product-oriented environmental policy) (Jackson 1999), greening of the supply chain (Sarkis 1995) and extended producer responsibility (EPR) (Lifset 1993).

Analysis of industrial or societal metabolism, that is, the tracking of materials and energy flows on a variety of scales is also motivated by a system orientation. Here reliance of research in industrial ecology on mass balances – making sure that inputs and outputs of processes add up in conformance with the first law of thermodynamics – reflects an effort at comprehensiveness. Because of the use of mass balances on these different scales, industrial ecology often involves the mathematics of budgets and cycles, and stocks and flows. By tracking chemical usage in a facility (Reiskin *et al.* 1999), nutrient flows in a city (Björklund *et al.* 1999), flows of heavy metals in river basins (Stigliani *et al.* 1994), or bulk materials in national economies (Adriaanse *et al.* 1997), industrial ecology seeks to avoid overlooking important uses of resources and/or their release to the environment. The tracking of materials and energy is sometimes embedded in the consideration of natural, especially biogeochemical, cycles and of how anthropogenic activities have perturbed those flows. For example, the study of anthropogenic perturbations of the nitrogen cycle is an important contribution of industrial ecology (Ayres, Schlesinger and Socolow 1994).

This same effort to examine human–environment interaction from a holistic perspective is manifested in formal systems modeling including dynamic modeling (Ruth and Harrington 1997), use of process models (Diwekar and Small 1998) and integrated energy, materials and emissions models such as MARKAL MATTER (2000) and integrated models of industrial systems and the biosphere (Alcamo *et al.* 1994). Such systems modeling not only increases the comprehensiveness of environmental analysis; it can also capture some of the interactions among the factors that drive the behavior of the system being studied (for example, Isaacs and Gupta 1997). Conceptual discussions of the nature of industrial ecology and sustainable development have highlighted the importance of non-linear behavior in human and environmental systems and argued that chaos theory and related approaches hold out potential for the field (Ruth 1996; Allenby 1999a), but little such work has been done to date.

Finally, the imperative for systems approaches is also reflected in a sympathy for the use of techniques and insights from multiple disciplines (Lifset 1998a; Graedel 2000). There have been some notable successes (Carnahan and Thurston 1998; van der Voet *et al.* 2000a), but multidisciplinary analysis – where several disciplines participate but not necessarily in an integrative fashion – is difficult and interdisciplinary analysis – where the participating disciplines interact and shape each other's approaches and results – is even more so. Interdisciplinarity remains an important challenge for not only industrial ecology, but all fields.

Technological Change

Technological change is another key theme in industrial ecology. It is a conspicuous path for pursuing the achievement of environmental goals as well as an object of study (Ausubel and Langford 1997; Grübler 1998; Norberg-Bohm 2000; Chertow 2001). In simple, if crude, terms, many in the field look to technological innovation as a central means of solving environmental problems. It should be noted, however, that while that impulse is shared widely within the field, agreement as to the degree to which this kind of innovation will be sufficient to solve technological problems remains a lively matter of debate (Ausubel 1996a; Graedel 2000).

Ecodesign (or design for environment – DFE) is a conspicuous element of industrial ecology (Chapter 36 of this handbook). By incorporating environmental considerations into product and process design *ex ante*, industrial ecologists seek to avoid environmental impacts and/or minimize the cost of doing so. This is technological innovation at the micro level, reflecting technological optimism and the strong involvement of academic and professional engineers. Ecodesign frequently has a product orientation, focusing on the reduction in the use of hazardous substances, minimization of energy consumption, or facilitation of end-of-life management through recycling and re-use. Implicitly, ecodesign relies on the life cycle perspective described earlier by taking a cradle-to-grave approach. Increasingly, it also strives for a systems approach, not only by considering impacts throughout the product life cycle, but also by employing comprehensive measures of environmental impact (Keoleian and Menerey 1994).

Ecodesign is complemented by research that examines when and how technological innovation for environmental purposes is most successful in the market (Preston 1997; Chertow 2000a). The focus on technological change in this field also has a macro version, examining whether technological change is good for the environment or how much change (of a beneficial sort) must be accomplished in order to maintain environmental quality. Here the IPAT equation (*I*mpact = *P*opulation × *A*ffluence × *T*echnology) has provided an analytical basis for parsing the relative contributions of population, economic growth (or, viewed in another way, consumption) and technology on environmental quality (Wernick, Waggoner and Ausubel 1997b; Lifset 2000, Chertow 2001). The equation provides a substantive basis for discussion of questions of carrying capacity implicit in the definition of industrial ecology offered earlier.

Role of Companies

Business plays a special role in industrial ecology in two respects. Because of the potential for environmental improvement that is seen to lie largely with technological innovation, businesses as a locus of technological expertise are an important agent for accomplishing environmental goals. Further, some in the industrial ecology community view command-and-control regulation as importantly inefficient and, at times, as counterproductive. Perhaps more significantly, and in keeping with the systems focus of the field, industrial ecology is seen by many as a means to escape from the reductionist basis of historic command-and-control schemes (Ehrenfeld 2000a). Regardless of the premise, a heightened role for business is an active topic of investigation in industrial ecology and a necessary component of a shift to a less antagonistic, more cooperative and, what is hoped, a more effective approach to environmental policy (Schmidheiny 1992).

This impulse to view business as a 'policy-maker rather than a policy-taker' (Socolow 1994, p.12) is reflected in a diverse set of analyses and initiatives that explore the efficacy of beyond-compliance environmental strategies and behavior. These include product take-back (Davis 1997), microeconomic rationales for beyond-compliance behavior (Reinhardt 1999), corporate environmental innovation pursued to maintain autonomy (Sharfman *et al.* 1998), corporate strategy and sustainable development (Hart and Milstein 1999) and macro-level analyses of the effectiveness of voluntary policy schemes (Harrison 1998).

Dematerialization and Eco-efficiency

Moving from a type I to a type II or III ecosystem entails not only closing loops, but using fewer resources to accomplish tasks at all levels of society. Reducing resource consumption and environmental releases thus translates into a cluster of related concepts: dematerialization, materials intensity of use, decarbonization and eco-efficiency (see Chapters 17 and 18). Dematerialization refers to the reduction in the quantity of materials used to accomplish a task; it offers the possibility of decoupling resource use and environmental impact from economic growth. Dematerialization is usually measured in terms of mass of materials per unit of economic activity or per capita and typically assessed at the level of industrial sectors, regional, national or global economies (Wernick, Herman, Govind and Ausubel 1997; Adriaanse *et al.* 1997). Decarbonization asks the analogous question about the carbon content of fuels (Nakicenovic 1997). Inquiry in this arena ranges from analysis of whether such reductions are occurring (Cleveland and Ruth 1998), whether dematerialization per se (that is, reduction in mass alone) is sufficient to achieve environmental goals (Reijnders 1997) and what strategies would be most effective in bringing about such outcomes (Weizsäcker *et al.* 1997). The intersection between investigation of dematerialization on the one hand, and other elements of industrial ecology such as industrial metabolism with its reliance on the analysis of the flows of materials on the other is clear. There is also overlap with industrial ecology's focus on technological innovation. This is because investigations of dematerialization often lead to questions about whether, at the macro or sectoral level, market activity and technological change autonomously bring about dematerialization (Cleveland and Ruth 1998) and whether dematerialization, expressed in terms of the IPAT equation, is sufficient to meet environmental goals.

At the firm level, an analogous question is increasingly posed as a matter of eco-efficiency, asking how companies might produce a given level of output with reduced use of environmental resources (Fussler 1996; OECD 1998b; DeSimone and Popoff 2000). Here, too, the central concern is expressed in the form of a ratio: output divided by environmental resources (or environmental impact). The connection between this question and industrial ecology's focus on the role of the firm and the opportunities provided through technological innovation is conspicuous as well.

Forward-looking Analysis

One final element of this field is worth noting. Much of research and practice in industrial ecology is intentionally prospective in its orientation. It asks how things might be done differently to avoid the creation of environmental problems in the first place, avoiding irreversible harms and damages that are expensive to remedy. Ecodesign thus plays a key role in its emphasis on anticipating and designing out environmental harms. More subtly, the field is optimistic about the potential of such anticipatory analysis through increased attention to system-level effects, the opportunities arising from technological innovation and from mindfulness of need to plan and analyze in and of itself. This does not mean that history is ignored. Industrial metabolism, for example, pays attention to historical stocks of materials and pollutants and the role that they can play in generating fluxes in the environment (Ayres and Rod 1986). However, industrial ecology does not emphasize remediation as a central topic in the manner of much of conventional environmental engineering.

Putting the Elements Together

There are (at least) two ways in which these themes and frameworks can be integrated into a larger whole. One is to view industrial ecology as operating at a variety of levels (Figure

1.2): at the firm or unit process level, at the inter-firm, district or sector level and finally at the regional, national or global level. While the firm and unit process is important, much of industrial ecology focuses at the inter-firm and inter-facility level, in part, as described above, because a systems perspective emphasizes unexpected outcomes – and possibly environmental gains – to be revealed when a broader scope is used and because pollution prevention, a related endeavor, has already effectively addressed many of the important issues at the firm, facility or unit process level.

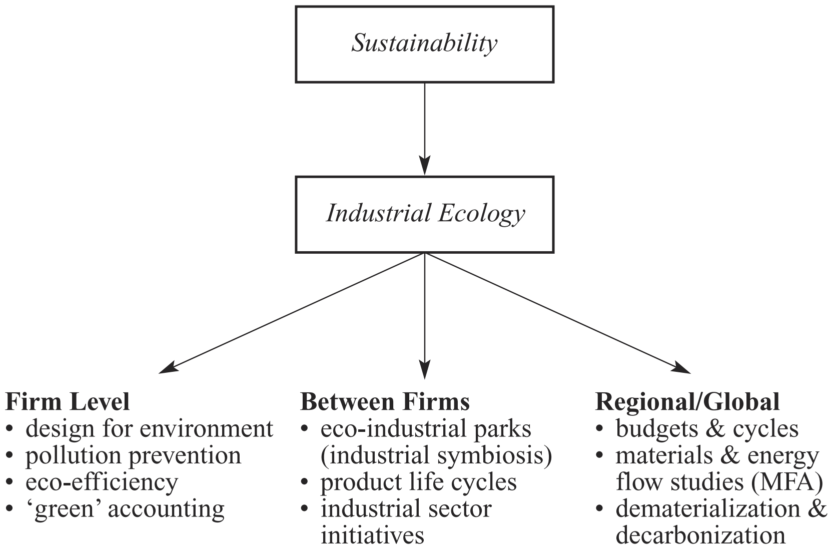

Figure 1.2 The elements of industrial ecology seen as operating at different levels

Another way to tie the elements together is to see them, as in Figure 1.3, as reflecting the conceptual or theoretical aspects of industrial ecology on the one hand and the more concrete, application-oriented tools and activities on the other. In this framework, many of the conceptual and interdisciplinary aspects of the field comprise the left side of the figure, while the more practical and applied aspects appear on the right side.

THE GOALS OF INDUSTRIAL ECOLOGY

Given this overview of the elements of industrial ecology, it is possible to entertain more complicated questions about this field. One set of especially notable and knotty questions revolve around the goals of industrial ecology. Clearly, the field is driven by concerns about human impact on the biophysical environment. Put simplistically, the goal is to improve and maintain environmental quality. Just as clearly, such a statement of goals does not begin to speak to the multiple dimensions of the research or practice in this field.

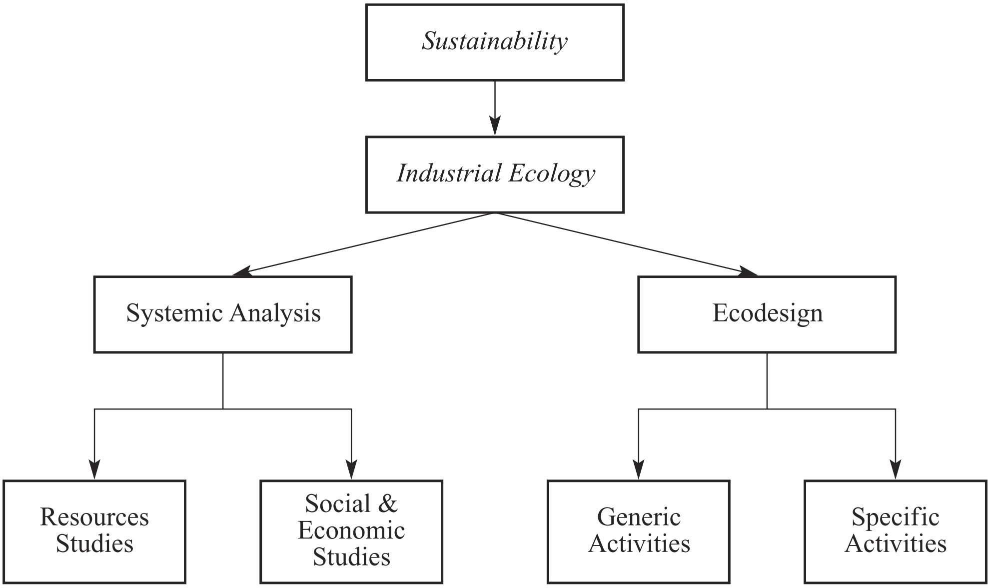

Figure 1.3 Industrial ecology conceptualized in terms of its system-oriented and application-oriented elements

Reducing Risk versus Optimizing Resource Use

Industrial ecology emphasizes the optimization of resource flows where other approaches to environmental science, management and policy sometimes stress the role of risk. For example, pollution prevention (P2) (also known as cleaner production or CP) emphasizes the reduction of risks, primarily, but not exclusively, from toxic substances at the facility or firm level (Allen 1996). Underlying this focus is an argument that only when the use of such substances is eliminated or dramatically reduced can the risks to humans and ecosystems be reliably reduced. In contrast, industrial ecology takes a systems view that typically draws the boundary for analysis more broadly – around groups of firms, regions, sectors and so on – and asks how resource use might be optimized, where resource use includes both materials and energy (as inputs) *and* ecosystems and biogeochemical cycles that provide crucial services to humanity (Ayres 1992a). In concrete terms, this means industrial ecology will sometimes look to recycling where P2 will emphasize prevention (Oldenburg and Geiser 1997). The differences between industrial ecology and P2 are not irreconcilable either conceptually or practically (van Berkel *et al.* 1997). In conceptual terms, P2 can be seen as a firm-level approach that falls under the broader rubric of industrial ecology (as shown in Figure 1.2). In concrete terms, the difference in actual practices by operating entities may not be great, although careful empirical work documenting how these two frameworks have differed in shaping decision making has not been conducted. However, some interesting analysis has been conducted of the risks posed by the recycling of hazardous materials, asking whether it is indeed possible to recycle such substances in an environmentally acceptable manner (Socolow and Thomas 1997; Karlsson 1999).

This is not the only way in which industrial ecology differs from allied fields in its orientation towards risk. The focus of industrial ecology on the flows of anthropogenic materials and energy is not often carried further than the point of release of pollutants into the environment. In contrast, much of traditional environmental science focuses precisely on the stages that follow such release – assessing the transport, fate and impact on human and non-human receptors. Similarly, risk assessment and environmental economics focus on the damages to humans and ecosystems, only sometimes looking upstream to the source of pollutants and the human activities that generate them. In this respect, industrial ecology can be seen as providing a complementary emphasis to these fields by concentrating on detailed and nuanced characterization of the sources of pollution. In a related vein, research in industrial ecology often examines perturbations to natural systems, especially biogeochemical cycles, arising from anthropogenic activities. The impacts of such perturbations can be construed in terms of risks to human health and economic well-being as well as to ecosystems, but the analysis of perturbations differs from the manner in which risk assessment – typically focused on threats to human health – is often conducted. This is not to suggest that industrial ecology ignores questions of risk, fate and transport or environmental endpoints. The intense work on methodologies for life cycle impact assessment (Udo de Haes 1996) is but one example of the field's efforts to systematically incorporate questions of environmental impact. Further, there is work in the field that integrates fate and transport into such analyses (Potting *et al.* 1998; Scheringer *et al.* 1999).

Another aspect of the focus on flows and releases rather than damages and endpoints is that the threats posed by releases – especially of persistent pollutants – endure and the receptors can change in a manner that later causes harms that may not be captured in a typical risk assessment. For example, cadmium deposition to agricultural soils that takes place as a result of naturally occurring cadmium contamination of phosphate fertilizers may not cause significant human health or ecological damage as long as fields are limed and thereby kept alkaline. If the fields are taken out of production, liming is likely to end. Soil pH will thereby increase, and cadmium may become biologically available and environmentally damaging (Stigliani and Anderberg 1994; Chapter 40).

Positive and Normative Analysis

One apparent tension related to the goals of industrial ecology relates to whether the field is positive (descriptive) or normative (prescriptive). If it is positive, then industrial ecology seeks to describe and characterize human–environment interactions, but not necessarily to alter them. On the other hand, if industrial ecology is normative, then some degree of human or environmental betterment is intrinsic to the goals of the field. This tension is reflected in multiple meanings accorded to key terms in the field. For example, the phrase 'industrial ecosystem' refers to facilities or industries that interact in a biophysical sense. Often it is a label for industrial districts like Kalundborg, where residuals are exchanged among co-located businesses. Leaving aside an especially loose usage that denotes any group of facilities, firms or industries, the question arises as to whether an industrial ecosystem necessarily refers to a desirable arrangement – where, for example, the participating firms extensively exchange residuals and thereby minimize releases of pollutants into the environment – or to a neutral description of a network of firms which might constitute either good industrial ecosystems (with considerable closing of loops and little pollution) or bad industrial ecosystems (with linear flows of resources and large amounts of pollution)[2]. The point is not the ambiguity in the terminology, but the difference in the emphasis that it reflects. Are there desirable end states that are integral to the notion of industrial ecology? Can one be an 'industrial ecologist' if one does not think that, more often than not, the closing of materials loops brings about environmental improvement? Is it necessary to think that the environmental situation is quite severe to engage in research in or the application of industrial ecology?

The ambiguity in general of the boundary between positive and normative endeavors plays a role here. Clearly, some fields exist at one pole or the other. Physics seeks to describe the physical universe without reference to ethical or other prescriptive principles. Theology, on the other hand, is obviously concerned with what ought to be. Economics, like many social sciences, represents a complex middle ground. On the one hand, it conceives of itself as value-neutral and engaged in the study of markets and the allocation of scarce resources. Yet, on the other, the field frequently offers advice of the sort 'if the goal is x, then the appropriate choice is y'. In this conception, the assertion of the value or importance of 'x' originates from outside the discipline, maintaining the apparent value neutrality of the analysis. But economists typically argue that the goal should be maximization of utility (at the individual level) and of social welfare (at the societal level) in a fashion that sharply constrains what 'x' might be and carrying with it an implicit set of value choices. Further, the analysis that economics uses to deduce y from x is argued by many to be value-laden (see, for example, Frank, Gilovich and Regan 1993).

Yet this tension may be over-stated. Contemporary engineering sciences fuse the positive and the normative without destroying the distinctiveness of each (Ehrenfeld 2000a). In practice, most industrial ecologists appear to enter the field out of concern for the potential environmental implications of production and consumption and the opportunities for improvement. It is thus regarded by most practitioners – those operating on the right side of Figure 1.3 – as a normative endeavor, one that is informed by the positive analysis generated by the activities on the left side of that figure. This characterization does not resolve all disputes about the normative status of field (Allenby 1999d; Boons and Roome 2000), but it does, we think, narrow the purview of the disagreements to questions about whether the field has teleological (i.e. goal-oriented) characteristics (Ehrenfeld 1997, 2000b) and to matters of meta-analysis such as by whom and, by what criteria, these sorts of debates are decided (Boons and Roome 2000).

Transformative and Incremental Change

Once some degree of normative content in the field is acknowledged, it is easy to entertain a related question of goals: is the environmental improvement that is sought large, transformative and discontinuous or is it incremental and continuous with current practice and infrastructure? Much of the conceptual discussion in industrial ecology looks to transformative change through the development and/or implementation of radically innovative technology, changes in consumption patterns, or new organizational arrangements. This type of transformative change can range from shifts to a hydrogen (Marchetti 1989) or carbohydrate economy (Morris and Ahmed 1992), factor 4 or factor 10 reductions in materials throughput (Weizsäcker *et al.* 1997), a shift to the use of services in lieu of products (Stahel 1994; Mont 2000) or new political–economic structures. At the same time, much of the practical work in ecodesign and life cycle management aims at more modest changes in product design protocols, materials choice, inter-firm relationships or environmental policy. Still other analyses assert that current arrangements meet the test of industrial ecology, employing the practices characteristic of the field, and need no significant improvement (Linden 1994). Yet the tension between transformative and incremental change may be overdrawn to the extent that, in many circumstances, the two paths are not mutually exclusive: the more modest changes can be pursued while the more ambitious ones are debated, refined and implemented. To the extent that such tensions do exist, they frequently reflect differing assessments of the severity of the current environmental situation as well as ideological differences about the degree to which market economies and current political institutions can and do achieve environmental goals.

THE BOUNDARIES OF INDUSTRIAL ECOLOGY

Like the question of goals, the boundaries of the field are subject to varying interpretation. They cannot be defined deductively from first principles; there is no authoritative epistemology in industrial ecology. At the same time, views on what are part of or outside this field shape, what is published in the *Journal of Industrial Ecology,* what is included in compendia such as this handbook and what projects receive funding from sources focusing on the development of industrial ecology.

One of the considerations with respect to the boundaries is whether industrial ecology should address not only 'what' but 'how' (Andrews 2000). Investigations of 'what' inform our understanding of the character of technological and natural systems – characterizing the manner in which these systems behave and interact, and under what circumstances environmental situations that humans deem preferable (for example, the absence of ozone holes in the atmosphere) might occur, much as in the definition of industrial ecology quoted in the beginning of this chapter. In that respect, the 'what' investigations include the what-if questions described above with respect to the normative aspects of industrial ecology: 'What if different materials were used for packaging; would carbon dioxide emissions decrease and global warming slow?' Some industrial ecologists, however, argue that the field must also embrace social, political and economic questions of 'how'. That is, given the identification of a preferred outcome, what strategies should be employed to bring about that outcome (Andrews 2000; Jackson and Clift 1998).

Thus the 'how' questions are largely a province of the social sciences. Sociology, economics, anthropology, psychology, political science and related fields have the potential to help identify strategies that are more likely to succeed. Nonetheless, the social sciences are not confined to 'how' questions (Fischoff and Small 1999). They can also indicate what is happening as when, for example, social scientists investigate the quantity and character of consumption in households and how it drives production and waste management activities and therefore environmental outcomes (Duchin 1998; Noorman and Schoot Uiterkamp 1998).

Most industrial ecologists would agree that such knowledge is crucial, but some would argue that that knowledge should remain lodged in allied fields, otherwise the boundaries and identity of industrial ecology will become so expansive as to be diffuse. Further complicating this tension are questions of whether these different sorts of inquiry can be construed as modular. That is, can they be pursued independently and *subsequently* melded to generate reliable insights? Or does their intellectual and organizational separation inevitably mean that the modular inquiries will be impoverished, incapable of integration, or even fundamentally misleading (Lifset 1998b)? Put more simply, *must* the questions that industrial ecology seeks to answer be pursued on an interdisciplinary basis to produce reliable answers? Ultimately, it will be the productivity of the various approaches in generating conceptual insights and practical knowledge that will determine their adoption[3].

CONCLUSION

As a new field, industrial ecology is a cluster of concepts, tools, metaphors and exemplary applications and objectives. Some aspects of the field have well-defined relationships, whereas other elements are only loosely grouped together, connected as much by the enthusiasm of the proponents as by a well-articulated intellectual architecture. We do not see this looseness as a fatal flaw in an emerging field, but rather as an opportunity for creativity and constructive discourse, and as a challenge.

NOTES

1. We put ‘natural’ in quotation marks because there are many ways in which the notion of natural ecosystems is complicated or contested. Many analysts argue that there are no longer any ecosystems unaffected by humankind, although clearly, even in this view, there is wide variation in the degree to which human activity dominates non-human ecosystems. More subtly, the notion of ‘natural’ is socially constructed and subject to varying interpretations across cultures (Williams 1980; Cronon 1996).
2. Multiple meanings extend to other terms in the field. ‘Industrial ecology’ is variously used to mean (a) a field of study, (b) a set of environmentally desirable practices and (c) the same practices as in meaning (b), but viewed neutrally. Such plurality of meanings is not unusual, however: ‘history’ refers both to past events and to the discipline that systematically studies those events.
3. Disagreement about industrial ecology’s boundaries are exacerbated by more pedestrian conflation of the ethics and values, the social sciences and public policy analysis. In particular, non-social scientists sometimes do not realize that the social sciences have a primarily positive/analytical focus, characterizing how humans behave, whereas it is the humanities that investigate and debate matters of values. Public policy analysis is often instrumental, asking how effectively certain strategies accomplish a set of public goals. Few industrial ecologists would suggest that the field offers powerful tools for adjudicating disputes over values, even if those disputes are important to the field.

2. Exploring the history of industrial metabolism

Marina Fischer-Kowalski

The scholarly influx of 'industrial metabolism' can be traced back for more than 150 years, across various scientific traditions, and even beyond the scope of industrial societies. In industrial ecology the term 'metabolism' is often treated as a metaphor, but earlier authors used this concept as a core analytical tool to develop an understanding of the energetic and material exchange relations between societies and their natural environments from a macro perspective. Several authors also displayed an explicit interest in human history as a history of changes in societies' metabolism.

The application of the term 'metabolism' to human society inevitably cuts across the 'great divide' (C.P. Snow) between natural and social sciences respectively. In the 1860s, when this divide was less rigid, the concept of metabolism from biology quickly found resonance in much of classic social science theory. Later on the social science use of this concept became more restricted.

The awakening of environmental awareness and the first skeptical views of economic growth during the late 1960s triggered a revival of interest in society's metabolism under the new perspective (Wolman 1965; Boulding 1966; Ayres and Kneese 1968a, 1969; Neef 1969; Boyden 1970; Georgescu-Roegen 1971; Meadows *et al.* 1972; Daly 1973). This survey ends with a brief mention of recent pioneering attempts to link IE with policy concerns.[1]

METABOLISM IN BIOLOGY, AGRONOMY AND ECOLOGY

A standard textbook in biology states

> to sustain the processes of life, a typical cell carries out thousands of biochemical reactions each second. The sum of all biological reactions constitutes metabolism. What is the purpose of these reactions – of metabolism? Metabolic reactions convert raw materials, obtained from the environment, into the building blocks of proteins and other compounds unique to organisms. Living things must maintain themselves, replacing lost materials with new ones; they also grow and reproduce, two more activities requiring the continued formation of macromolecules. (Purves *et al.* 1992, p. 113)

Or, later:

> Metabolism is the totality of the biochemical reactions in a living thing. These reactions proceed down metabolic pathways, sequences of enzyme catalyzed reactions, so ordered that the product

> of one reaction is the substrate for the next. Some pathways synthesize, step by step, the important chemical building blocks from which macromolecules are built, others trap energy from the environment, and still others have functions different from these. (Ibid., p. 130)

The term 'metabolism' (Stoffwechsel) was introduced as early as 1815 and was adopted by German physiologists during the 1830s and 1840s to refer primarily to material exchanges within the human body, related to respiration. Justus Liebig then extended the use of the term to the context of tissue degradation, in combination with the somewhat mystic notion of 'vital force' (Liebig 1964 [1842]). By contrast, Julius Robert Mayer, one of the four co-discoverers of the law of conservation of energy, criticized the notion of 'vital force' and claimed that metabolism was explicable entirely in terms of conservation of energy and its exchange (Mayer 1973 [1845]). Later this term was generalized still further and emerged as one of the key concepts in the development of biochemistry, applicable both at the cellular level and in the analysis of entire organisms (Bing 1971).

Whereas the concept of metabolism was and still is widely applied at the interface of biochemistry and biology when referring to cells, organs and organisms, it became a matter of dispute whether it is applicable on any level further up the biological hierarchy. E.P. Odum clearly favors the use of terms like 'growth' or 'metabolism' on every biological level from the cell to the ecosystem (for example, Odum 1959). Which processes may and should be studied on hierarchical levels beyond the individual organism, though, is a matter of debate dating back to Clements (1916) and still going on.

Tansley (1935) established 'ecosystem' as a proper unit of analysis. He opposed Clements' 'creed' of an organismical theory of vegetation. Lindemann (1942) analyzed ecosystems mathematically, with plants being the *producer* organisms to convert and accumulate solar radiation into complex organic substances (chemical energy) serving as food for animals, the *consumer* organisms of ecosystems. Every dead organism then is a potential source of energy for specialized *decomposers* (saprophagous bacteria and fungi) thereby closing the cycle. This is in essence what Odum referred to when talking about the metabolism in an ecosystem.

Basically this is a debate about 'holism' (or organicism) v. 'reductionism'. Do populations (that is, the interconnected members of a species), communities (the total of living organisms in an ecosystem) or ecosystems (the organisms and the effective inorganic factors in a habitat) have a degree of systemic integration comparable to individual organisms? Does evolution work upon them as units of natural selection? These questions are contested in biology, and thus a use of the term 'metabolism' for a system consisting of a multitude of organisms does not pass unchallenged.

Like any other animal, humans are heterotrophic organisms, drawing energy from complex organic compounds (foodstuff) that have been directly or indirectly synthesized by plants from air, water and minerals, utilizing the radiant energy from the sun. But humans as a species are not able to survive and maintain their metabolism individually. Does it make sense, therefore, to look at human communities and societies in terms of entities of cooperatively empowered metabolism? Societies will be bound to have collective metabolism that is, at least, the sum of the individual metabolisms of its members. If a society cannot maintain this metabolic turnover, its population will die or emigrate. But not all materials need to be processed through the cells of human bodies. From an ecosystem perspective, for example, the materials birds use in building their nests constitute a material flow associated with birds. In ordinary biological language, however, this is not part of bird metabolism, although it may be vital for bird reproduction. Thus the concept 'metabolism' needs to be expanded to encompass material and energetic flows and transformations associated with 'living things' but extending beyond the anabolism and catabolism of cells. The overall material and energetic turnover of an ecosystem component, its consumption of certain materials, their transformation and the production of other materials, may be an ecologically useful parameter. In biology this would not be called metabolism.

Humans, of course, sustain at least part of their metabolism not by direct exchanges with the environment (as, for example, in breathing), but via the activities of other humans. This is a matter of social organization. Any attempt to describe this organization in terms of a biological system – whether it be the organism, or a population in a habitat, or an ecosystem – draws on analogies and runs the risk of being reductionist.[2] On the other hand, the concept of metabolism in biology has valuable features: It refers to a highly complex self-organizing process which the system seeks to maintain in widely varying environments. This metabolism requires certain material inputs from the environment, and it returns these materials to the environment in a different form.

METABOLISM IN THE SOCIAL SCIENCES

Metabolism in Social Theory

It was Marx and Engels who first applied the term 'metabolism' to society. 'Metabolism between man and nature' is used in conjunction with their basic, almost ontological, description of the labor process.

> Labor is, first of all, a process between man and nature, a process by which man, through his own actions, mediates, regulates and controls the metabolism between himself and nature. He confronts the materials of nature as a force of nature. He sets in motion the natural forces which belong to his own body, his arms, legs, head and hands, in order to appropriate the materials of nature in a form adapted to his own needs. Through this movement he acts upon external nature and changes it, and in this way he simultaneously changes his own nature . . . [the labor process] is the universal condition for the metabolic interaction [Stoffwechsel] between man and nature, the everlasting nature imposed condition of human existence. (Marx 1976 [1867], pp. 283, 290)

According to Foster (1999, 2000), and in contrast to earlier interpretations (Schmidt 1971), Marx derived much of his understanding of metabolism from Liebig's analysis of nutrient depletion of the soil following urbanization.

> Large landed property reduces the agricultural population to an ever decreasing minimum and confronts it with an ever growing industrial population crammed together in large towns; in this way it produces conditions that provoke an irreparable rift in the interdependent process of social metabolism, a metabolism prescribed by the natural laws of life itself. The result of this is a squandering of the vitality of the soil, which is carried by trade far beyond the bounds of a single country. (Liebig 1842, as quoted in Marx 1981 [1865], p. 949; similarly in Marx 1976 [1867]; see also Foster 2000, pp. 155f)

According to Foster, the concept of metabolism and 'metabolic rift' provided Marx with a materialist way of expressing his notion of alienation from nature that was central to his critique of capitalism from his earliest writings on.

> Freedom in this sphere [the realm of natural necessity] can consist only in this, that socialized man, the associated producers, govern the human metabolism with nature in a rational way, bringing it under their own collective control instead of being dominated by it as a blind power; accomplishing it with the least expenditure of energy and in conditions most worthy and appropriate with their human nature. (Marx 1981 [1865], p. 959)

Thus Marx employed the term 'metabolism' for the material exchange between man and nature on a fundamental anthropological level, as well as for a critique of the capitalist mode of production. But the accumulation of capital has nothing to do with the appropriation of the accumulated 'wealth' of nature (for example, fossil fuels); appropriation as a basis for capital accumulation is always and only appropriation of surplus human labor, as Martinez-Alier (1987, pp. 218–24) pointed out.

The writings of Marx and Engels are not the only reference to societal metabolism from the 'founding fathers' of modern social science. Most social scientists of those times were interested in evolutionary theory and its implications for universal progress (for example, Spencer 1862; Morgan 1877). Societal progress and the differences in stages of advancement among societies relate to the amount of available energy: societal progress is based on energy surplus. Firstly it enables social growth and thereby social differentiation. Secondly it provides room for cultural activities beyond basic vital needs (Spencer 1862).

Nobel Prize-winning chemist Wilhelm Ostwald argued that minimizing the loss of free energy is the objective of every cultural development. Thus one may deduce that the more efficient the transformation from crude energy into useful energy, the greater a society's progress (Ostwald 1909). For Ostwald the increase of energy conversion efficiency has the characteristics of a natural law affecting every living organism and every society. He stressed that each society has to be aware of the 'energetic imperative [Energetische Imperativ]': 'Don't waste energy, use it' (Ostwald 1912, p. 85). Ostwald was one of the few scientists of his time who was sensitive to the limitations of fossil resources. According to him, a durable (sustainable) economy must use solar energy exclusively. This work provided Max Weber (1909) with the opportunity for an extensive discussion. Weber reacted in quite a contradictory manner. On the one hand he dismissed Ostwald's approach as 'grotesque' (Weber 1909, p. 401) and challenged its core thesis on natural science grounds: 'In no way would an industrial production be more energy efficient than a manual one – it would only be more cost efficient' (ibid., pp. 386ff). At the same time he rejected natural science arrogance towards the 'historical' sciences and the packaging of value judgments and prejudices in natural science 'facts' (ibid., p. 401). On the other hand, although he admitted that energy may possibly be important to sociological concerns (ibid., p. 399; see also Weber (1958 [1904]), he never elaborated such considerations.

Sir Patrick Geddes, co-founder of the British Sociological Society, sought to develop a unified calculus based upon energy and material flows and capable of providing a coherent framework for all economic and social activity (Geddes 1997 [1884]). He proclaimed the emancipation from monetary economy towards an economy of energy and resources. In four lectures at the Royal Society of Edinburgh, Geddes developed a type of economic input–output table in physical terms. The first column contains the sources of energy as well as the sources for materials used. Energy and materials are transformed into products in three stages: the extraction of fuels and raw materials; the manufacture; and the transport and exchange. Between each of these stages there occur losses that have to be estimated – the final product might then be surprisingly small in proportion to the overall input (Geddes 1885). Far ahead of his time, Geddes appears to have been the first scientist to approach an empirical description of societal metabolism on a macroeconomic level.

Frederick Soddy, another Nobel laureate in chemistry, also turned his attention to the energetics of society, but did so with an important twist: he saw energy as a critical limiting factor to society and thus was one of the few social theorists sensitive to the second law of thermodynamics (Soddy 1912, 1922, 1926). He thereby took issue directly with Keynes' views on long-term economic growth, as appreciated by Daly (1980). In the mid-1950s, Fred Cottrell (1955) again raised the idea that available energy limits the range of human activities. According to him this is one of the reasons why pervasive social, economic, political and even psychological change accompanied the transition from a low-energy to a high-energy society.

For the development of sociology as a discipline these more or less sweeping energetic theories of society remained largely irrelevant. Even the influential Chicago-based school of sociology with the promising label 'human ecology' (for example, Park 1936) carefully circumvented any references to natural conditions or processes. Before the advent of the environmental movement, modern sociology did not refer to natural parameters as either causes or consequences of human social activities.

Metabolism in Cultural and Ecological Anthropology

The beginnings of cultural anthropology (as in the works of Morgan 1877) were, similar to sociology, marked by evolutionism, that is, the idea of universal historical progress from more 'natural', barbarian to more advanced and civilized social conditions. Then cultural anthropology split into a functionalist and a culturalist tradition. The functionalist line, from which contributions to societal metabolism should be expected, did not, as was the case in sociology, turn towards economics and distributional problems, but retained a focus on the society–nature interface. In effect, several conceptual clarifications and rich empirical material on societies' metabolism can be gained from this research tradition.

Leslie White, one of the most prominent anthropologists of his generation and an early representative of the functionalist tradition, rekindled interest in energetics. For White, the vast differences in the types of extant societies could be described as social evolution, and the mechanisms propelling it were energy and technology. 'Culture evolves as the amount of energy harnessed per capita and per year is increased, or as the efficiency of the instrumental means (i.e. technology) of putting the energy to work is increased' (White 1949, p. 366). A society's level of evolution can be assessed mathematically: it is the product of the amount of per capita energy times efficiency of conversion. So this in fact was a metabolic theory of cultural evolution – however unidimensional and disregarding of environmental constraints it may have been.

Julian Steward's 'method of cultural ecology' (Steward 1968) paid a lot of attention to the quality, quantity and distribution of resources within the environment. His approach can be illustrated from the early comparative study, *Tappers and Trappers* (Murphy and Steward 1955). Two cases of cultural (and economic) change are presented, in which tribes traditionally living from subsistence hunting and gathering (and some horticulture) completely change their ways of living. The authors analyze it as an irreversible shift from a subsistence economy to dependence upon trade.

Several outright analyses of metabolism have been produced by the 'neofunctionalists': Marvin Harris (1966, 1977), Andrew Vayda and Roy Rappaport (Rappaport 1971; Vayda and Rappaport 1968). The followers of this approach, according to Orlove (1980, p. 240), 'see the social organization and culture of specific populations as functional adaptations which permit the populations to exploit their environments successfully without exceeding their carrying capacity'. The unit which is maintained is a given population rather than a particular social order (as it is with sociological functionalists). In contrast to biological ecology, the neofunctionalists treat adaptation, not as a matter of individuals and their genetic success, but as a matter of cultures. Cultural traits are units which can adapt to environments and which are subject to selection.[3] In this approach, human populations are believed to function within ecosystems as other populations do, and the interaction between populations with different cultures is put on a level with the interaction of different species within ecosystems (Vayda and Rappaport 1968).

This approach has been very successful in generating detailed descriptions of food-producing systems (Anderson 1973; Kemp 1971; Netting 1981). In addition to that, it has raised the envy of colleagues by successfully presenting solutions to apparent riddles of bizarre habits and thereby attracting a lot of public attention (Harris 1966, 1977; Harner 1977).

There certainly are some theoretical and methodological problems in neofunctionalism which need to be discussed in greater detail. They entail the difficulty of specifying a unit of analysis: a local population? A culture? This is related to the difficulty of specifying the process of change, and to the difficulty of locating intercultural (or inter-society) interactions in this framework. These scientific traditions, however, have prepared cultural anthropologists to be among the first social scientists actively participating in the later discussion of environmental problems of industrial metabolism.

Metabolism in Social Geography and Geology

In 1955 a total of 70 participants from all over the world and from a great variety of disciplines convened in Princeton, New Jersey, for a remarkable conference: 'Man's Role in Changing the Face of the Earth'. The conference was financed by the Wenner-Gren Foundation for Anthropological Research. The geographer Carl O. Sauer, the zoologist Marston Bates and the urban planner Lewis Mumford presided over the sessions. The papers and discussions were published in a 1200-page compendium (Thomas 1956a) that perhaps documents the world's first high-level interdisciplinary panel on environmental problems of human development.

The title of the conference honored George Perkins Marsh, who published the book, *Man and Nature; or, Physical Geography as Modified by Human Action*, in 1864, and is considered the father of social geography. For Marsh, man was a dynamic force, often irrational in creating a danger to himself by destroying his base of subsistence. The largest chapter of *Man and Nature*, entitled 'The Woods', advocated the recreation of forests in the mid-latitudes. He was not, as the participants of the conference note, concerned about the exhaustion of mineral resources. He looked upon mining rather from an aesthetic point of view, considering it 'an injury to the earth' (Thomas 1956b, p. xxix).

The issue of possible exhaustion of mineral resources was taken up by the Harvard geologist Nathaniel Shaler in his book *Man and the Earth* (1905). In considering longer time series, he noted that 'since the coming of the Iron Age' the consumption of mineral resources had increased to a frightening degree. In 1600 only very few substances (mostly precious stones) had been looked for underground. But, at the turn of the 20th century, there were several hundred substances from underground sources being used by man, of essential importance being iron and copper. Shaler was concerned with the limits of the resource base.

This shift of focus from Marsh (1973 [1864]) to Shaler (1905) reflects the change in society's metabolism from an agrarian mode of production (where scarcity of food promotes the extension of agricultural land at the expense of forests) to an industrial one, where vital 'nutrients' are drawn from subterrestrial sinks that one day will be exhausted.

In Thomas (1956a) the concern with a limited mineral base for an explosively rising demand for minerals is even more obvious. Such a 'materials flow' focus seems to have been strongly supported by wartime experiences and institutions: Samuel H.J. Ordway quoted data from Paley (1952) – an excellent source for longer time series of materials consumption – worrying about the 'soaring demand' for materials.[4] The depletion of national resources becomes part of a global concern: 'If all the nations of the world should acquire the same standard of living as our own, the resulting world need for materials would be six times present consumption' (Ordway 1956, p. 988). Ordway advanced his 'theory of the limit of growth', based on two premises:

> 1. Levels of human living are constantly rising with mounting use of natural resources. 2. Despite technological progress we are spending each year more resource capital than is created. The theory follows: If this cycle continues long enough, basic resources will come into such short supply that rising costs will make their use in additional production unprofitable, industrial expansion will cease, and we shall have reached the limit of growth. (Ordway 1956, p. 992)

It is interesting to note that even the idea of materials' consumption growing less than GDP because of increases in efficiency was taken up: in its projections for 1975 the Paley Report expected US GDP to double compared to 1950, but the materials input necessary for this to rise by only 50–60 per cent (data from Ordway 1956, p. 989).

McLaughlin, otherwise more optimistic than Ordway, stated in the same volume that by 1950 for every major industrial power the consumption of metals and minerals had exceeded the quantity which could be provided from domestic sources (McLaughlin 1956, p. 860).

Similarly, the 1955 conference participants discussed the chances of severe shortages in future energy supply. Eugene E. Ayres, speaking of 'the age of fossil fuels', and Charles A. Scarlott, treating 'limitations to energy use', emphasized the limits inherent to using given geological stocks. Ayres, elaborating on fossil fuels since the first uses of coal by the Chinese about two thousand years ago, was very skeptical about geologists' estimates (then) of the earth's reserves, suspecting them of being vastly understated. He nevertheless concluded: 'In a practical sense, fossil fuels, after this century, will cease to exist except as raw materials for chemical synthesis' (Ayres 1956, p. 380). Scarlott (1956) demonstrated the diversification of energy uses and the accompanying rise in demand and then elaborated on a possible future of solar energy utilization and nuclear fusion as sources of energy.

In the 1955 conference materials flow considerations were mainly confined to the input side of societal metabolism. The overall systemic consideration that the mobilization of vast amounts of matter from geological sinks (for example, minerals and fossil energy carriers) into a materially closed system such as the biosphere would change parameters of atmospheric, oceanic and soil chemistry on a global level does not appear there. Still, many contributions to this conference document the transformations of local and regional natural environments by human activity, both historical and current. These concerns were also explicitly addressed in *The Earth as Transformed by Human Action: Global and Regional Changes in the Biosphere over the Past 300 Years* (Turner *et al.* 1990), representing the contemporary state of the art of social geography.

The global environmental change issue was taken up in a special issue of *Scientific American* in September 1970, devoted to the biosphere. One year later, *Scientific American* edited an issue on energy and socioeconomic energy metabolism. In 1969 the German geographer Neef explicitly talked about the 'metabolism between society and nature' as a core problem of geography (Neef 1969). But this already belongs to the post-1968 cultural revolution of environmentalism we will treat next.

THE PIONEERS OF ECONOMY-WIDE MATERIALS FLOW ANALYSIS IN THE LATE 1960S

In the late 1960s, when it became culturally possible to take a critical stance with respect to economic growth and consider its environmental side-effects, the stage was set for a new twist in looking at society's metabolism. Up to this point, metabolism had mainly come up in various arguments claiming that natural forces and physical processes did, indeed, matter for the organization and development of society, and that it would be reasonable, therefore, to attribute to them some causal significance for social facts. The mainstream of social science dealing with modern industrial society – be it economics, sociology or political science – had not cared about this issue at all. In the mid-1960s this started to change and, apparently originating from the USA, a set of new approaches were developed, often triggered by natural scientists, and subsequently further developed in cooperation with social scientists. In these approaches the material and energetic flows between societies (or economies) and their natural environment finally became a major issue. The common picture of cultural evolution as eternal progress started to give way to a picture of industrial economic growth as a process which possibly implied the fatal devastation of human life. This must be considered as quite a basic change in world views, and it took hold of a wide range of intellectuals across many disciplines. It promoted something like a rebirth of the paradigm of metabolism, applied to industrial societies.

> The metabolic requirements of a city can be defined as the materials and commodities needed to sustain the city's inhabitants at home, at work and at play. (. . .) The metabolic cycle is not completed until the wastes and residues of daily life have been removed and disposed of with a minimum of nuisance and hazard. (Wolman 1965, p. 179)

These lines served as the introduction to the first attempt to conceptualize and operationalize the metabolism of industrial society; that is, the case study of a model US city of one million inhabitants. Abel Wolman was well aware of the fact that water is by far the largest input needed, but he also offered estimates for food and fossil energy inputs, as well as (selected) outputs such as refuse and air pollutants. His argument was mainly directed at problems he foresaw concerning the provision of an adequate water supply for US megacities. A few years later an Australian team analyzed the metabolism of Hong Kong, concentrating on its 'biometabolism' (that is, human and animal nutrient cycles) only. A comparison with Sydney (data for the years 1970 and 1971) showed a 'Western-style' diet, with the same calorific and nutrient benefit for the consumer, to be about twice as wasteful as a diet in the Chinese tradition (Newcombe 1977; Boyden *et al.* 1981).

In *The Economics of the Coming Spaceship Earth* with reference to Bertalanffy (1952), Kenneth Boulding briefly outlined an impending change from what he called a 'cowboy economy' to a 'spaceman economy' (Boulding 1966). The present world economy, according to this view, is an open system with regard to energy, matter and information ('econosphere'). There is a 'total capital stock' (the set of all objects, people, organizations and so on that have inputs and outputs). Objects pass from the non-economic to the economic set in the process of production, and objects pass out of the economic set 'as their value becomes zero' (ibid., p. 5). 'Thus we see the econosphere as a material process.' This similarly can be described from an energetic point of view. In the 'cowboy economy', throughput is at least a plausible measure of the success of the economy.

> By contrast, in the spaceman economy, throughput is by no means a desideratum, and is regarded as something to be minimized rather than maximized. The essential measure of the success of the economy not its production and consumption at all, but the nature, extent, quality and complexity of the total capital stock, including in this the state of the human bodies and minds. (Ibid., p. 9)

In 1969 Robert Ayres, a physicist, and Allen Kneese, an economist, basically presented the full outline of what – much later, in the 1990s – was to be carried out as material flow analyses of national economies (Ayres and Kneese 1969). Their article was based upon a report prepared for the US Congress by a Joint Economic Committee and published in a volume of federal programs in 1968 (see Ayres and Kneese 1968a). Their core argument was an economic one: the economy draws heavily on priceless environmental goods such as air and water – goods that are becoming increasingly scarce in highly developed countries – and this precludes Pareto-optimal allocations in markets at the expense of those free common goods. They concluded with a formal general equilibrium model to take care of these externalities. In the first part of the paper the authors gave an outline of the problem and presented a first crude material flow analysis for the USA, 1963–5. They claimed 'that the common failure of economics [. . .] may result from viewing the production and consumption processes in a manner that is somewhat at variance with the fundamental law of the conservation of mass' (Ayres and Kneese 1969, p. 283). Uncompensated externalities must occur, they argued, unless either (a) all inputs of the production process are fully converted into outputs, with no unwanted residuals along the way (or else they are all to be stored on the producers' premises), and all final outputs (commodities) are utterly destroyed in the process of consumption, or (b) property rights are so arranged that all relevant environmental attributes are privately owned and these rights are exchanged in competitive markets.

> Neither of these conditions can be expected to hold. (. . .) Nature does not permit the destruction of matter except by annihilation with antimatter, and the means of disposal of unwanted residuals which maximizes the internal return of decentralized decision units is by discharge to the environment, principally watercourses and the atmosphere. Water and air are traditionally free goods in economics. But in reality . . . they are common property resources of great and increasing value. (. . .) Moreover (. . .) technological means for processing or purifying one or another type of waste discharge do not destroy the residuals but only alter their form. (. . .) Thus (. . .) recycle of materials into productive uses or discharge into an alternative medium are the only general options. (Ayres and Kneese 1969, p. 283).

> Almost all of standard economic theory is in reality concerned with services. Material objects are merely vehicles which carry some of these services . . . Yet we (the economists) persist in referring to the 'final consumption' of goods as though material objects . . . somehow disappeared into the void . . . Of course, residuals from both the production and consumption processes remain and they usually render disservices . . . rather than services. (Ibid., p. 284)

Thus they proposed to 'view environmental pollution and its control as a *materials balance problem* for the entire economy' (ibid., p. 284, emphasis added).

> In an economy which is closed (no imports or exports) and where there is no net accumulation of stocks (plant, equipment . . . or residential buildings), the amount of residuals inserted into the natural environment must be approximately equal to the weight of basic fuels, food, and raw materials entering the processing and production system, plus oxygen taken from the atmosphere. (Ibid.)

Within these few paragraphs, almost all elements of the future debate emerged. The model of socioeconomic metabolism presented (a term not used in the contribution) owes more to physics than to ecology. For an organism, it is obvious that some residues have to be discharged into the environment. In population ecology, it is the efficiency of energetic conversion that would be considered – not the recycling of materials. This clearly would be the task of the ecosystem: in the ecosystem it is the 'division of labor' of different species that would take care of materials recycling, and never the members of one species only. From the point of view of ecosystems theory, therefore, the idea of residues as a 'disservice' to the population discharging them would seem alien to the common concept of nutrient cycles.

Whereas the inputs from the environment to the economy were listed in some detail, the outputs to the environment (in the sense of residuals) were only treated in a sweeping manner. Nevertheless, all the problems that have marked the following decades of emission (and waste policies) – problems that still have not been properly resolved – were clearly set forth. It was explicitly stated that there is a primary interdependency among all waste streams that evades treatment by separate media. The authors of this article even recognized that there is one stream of waste that is non-toxic and, hence, not interesting for emission regulation: carbon dioxide. They anticipated correctly that carbon dioxide, by reason of its sheer quantity, might become a major problem in changing the climate. Finally, they were able to see that a reduction of residuals can only be achieved via a reduction of inputs. In a sense, they could be said to have 'invented' all these core insights into the materials balance approach. This contribution became the starting point of a research tradition capable of portraying the material and energetic metabolism of advanced industrial economies. It was not 'man' any more that was materially and energetically linked to nature, but a complex and well-defined social system: 'The dollar flow governs and is governed by a combined flow of materials and services (value added)' (Kneese *et al.* 1974, p. 54).

Judged by the standards of later empirical analyses (for example, Adriaanse *et al.* 1997; Matthews *et al.* 2000), the empirical results rendered by these pioneering studies appear to be correct within an order of magnitude: they arrive at about 20 metric tons per capita population and year as 'direct material input' into the economy. (For details see Fischer-Kowalski 1998, pp. 71ff.)

We may conclude, therefore, that the pioneer studies of economy-wide material metabolism not only set up an appropriate conceptual framework, but also arrived at reasonable empirical results. Considering this fact, it is amazing that it took another 20 years for this paradigm and methodology to become more widely recognized as a useful tool.

NOTES

1. The period since the 1960s, in which there has been a virtual explosion of research dealing with industrial metabolism, is the subject of a review in the *Journal of Industrial Ecology* (Fischer-Kowalski and Hüttler 1999). For an excellent review covering the history of economics, see Martinez-Alier (1987).
2. It is interesting to note that biologists tend to attribute organismic (or system integration) characteristics to the human society where they might deny them to an ecosystem. For an early example see Tansley (1935, p. 290).
3. While cultural maladaptation to an environment may in fact harm the population concerned, it will not as a rule systematically change its genetic composition. If as a consequence cultural changes occur, they will most likely be the result of learning.
4. Ordway (1956, p. 988) even constructed a number for the 'raw material consumption' of the USA in 1950; '2.7 billion tons of materials of all kinds – metallic ores, nonmetallic minerals, construction materials and fuels'. Ayres and Kneese (1969) gave this consumption as 2.4 billion tons (including agricultural products, but excluding construction materials).

3. The recent history of industrial ecology

Suren Erkman*

Industrial ecology has been manifest intuitively for quite a long time. In the course of the past 30 years the several attempts made in that direction mostly remained marginal. The expression re-emerged in the early 1990s, at first among a number of industrial engineers connected with the National Academy of Engineering in the USA.

So far, there is no standard definition of industrial ecology, and a number of authors do not make a clear difference between industrial metabolism and industrial ecology. The distinction, however, makes sense not only from a methodological point of view, but also in a historical perspective: the 'industrial metabolism' analogy was currently in use during the 1980s, especially in relation to the pioneering work of Robert Ayres, first in the US, then at the International Institute for Applied Systems Analysis (IIASA, Laxenburg, Austria) with William Stigliani and colleagues, and more recently at INSEAD (Fontainebleau, France) (Ayres and Kneese 1969; Ayres 1989a, 1989b, 1992b, 1993b; Ayres *et al.* 1989; Stigliani and Jaffé 1993; Stigliani and Anderberg 1994; Lohm *et al.* 1994). At about the same time, the metabolic metaphor was pursued independently by Peter Baccini, Paul Brunner and their colleagues at the Swiss Federal Institute of Technology (ETHZ) (Baccini and Brunner 1991; Brunner *et al.* 1994). In parallel, it should be recalled that there is a long tradition of organic metaphors in the history of evolutionary economics (Hodgson 1993a, 1993b).

INDUSTRIAL ECOLOGY: EARLIER ATTEMPTS

There is little doubt that the concept of industrial ecology existed well before the expression, which began to appear sporadically in the literature of the 1970s. As usual, on certain occasions the same expression does not refer to the same concept. In the case of industrial ecology, it was referring to the regional economic environment of companies (Hoffman 1971; Hoffman and Shapero 1971) or was used as a 'green' slogan by some industrial lobbies in reaction to the creation of the United States Environmental Protection Agency (US EPA) (Gussow and Meyers 1970). On the other hand, the concept of industrial ecosystems is clearly present, although not explicitly named, in the writings of systems ecologists such as Odum and Hall (Odum and Pinkerton 1955; Hall 1975). In fact, and not surprisingly, systems ecologists studying biogeochemical cycles had for a very long time the intuition of the industrial system as a subsystem of the biosphere

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(Hutchinson 1948; Brown 1970). But this line of thought has never been actively investigated, with the notable exception of agroecosystems, whereas the recent industrial ecology perspective acknowledges the existence of a wide range of industrial ecosystems with varying degrees and patterns of interactions with the biosphere, from certain kinds of almost 'natural' agroecosystems to the supremely artificial ecosystems, like space ships (Cole and Brander 1986; Jones *et al.* 1994; Folsome and Hanson 1986; Lasseur 1994).

What might be the earliest occurrence of the expression 'industrial ecosystem' (in accordance with today's concept and in the published literature in English) can be found in a paper by the late well-known American geochemist, Preston Cloud. This paper was presented at the 1977 Annual Meeting of the German Geological Association (Cloud 1977). Interestingly, it is dedicated to Nicholas Georgescu-Roegen, the pioneer of bioeconomics who on many occasions has insisted on the importance of matter and material flows in the human economy in a thermodynamical perspective, and has also extensively written on technological dynamics (Georgescu-Roegen 1976a, 1976b, 1978, 1979a, 1979b, 1980, 1982, 1983, 1984, 1986, 1990; Grinevald, 1993).

Several attempts to launch this new field have been made in the last couple of decades, with very limited success. Charles Hall, an ecologist at New York State University, began to teach the concept of industrial ecosystems and publish articles on it in the early 1980s, without getting any response (Hall *et al.* 1992). At about the same time, in Paris, another academic, Jacques Vigneron, independently launched the notion of industrial ecology, without awakening any real interest for his part, either (Vigneron 1990).

The industrial ecology concept was indisputably in its very early stages of development in the mid-1970s, in the context of the flurry of intellectual activity that marked the early years of the United Nations Environment Program (UNEP). Set up following the 1972 United Nations Conference on Human Environment in Stockholm, UNEP's first director was Maurice Strong. One of his close collaborators at the time was none other than Robert Frosch, who was to make a decisive contribution to the revival of the concept of industrial ecology thanks to an article published in 1989 in the monthly magazine *Scientific American*.

A similar intellectual atmosphere also prevailed around the same period in other circles, such as the United Nations Industrial Development Organization (UNIDO) and the United Nations Economic Commission for Europe (ECE). For example, many papers presented during an international seminar organized by the ECE in 1976 on what was called at that time 'Non-waste Technology and Production' disseminated ideas similar to those discussed today in the cleaner production and industrial ecology literature (ECE 1978). Another example: Nelson Nemerow, who has been active in the industrial waste treatment field in the USA for more than 50 years, acknowledges in a his book a brainstorming session with Alex Anderson of UNIDO in Vienna during the early 1970s, at which time the idea of 'environmentally balanced industrial complexes' in the perspective of zero pollution was born (Nemerow 1995). Very similar ideas were discussed by Theodore Taylor, a nuclear physicist turned environmentalist, and Charles Humpstone, a lawyer, in a book published in New York at around the same time (Taylor and Humpstone 1972). In fact, in 1967 Taylor created the International Research and Technology Corporation (IR&T, based in Washington, DC), a company devoted to the development of these concepts, of which he was the president and Robert Ayres the vice-president. Clearly, ideas such as 'environmentally balanced industrial complexes' proposed in the early 1970s can be considered as precursors of more recent concepts, like eco-industrial parks (Lowe 1992) and zero emission industrial clusters (Pauli 1995). New other examples of similar thinking could be provided, and it is likely that many of them have not yet been documented (Farvar and Milton 1972; Dasmann *et al.* 1973). This is especially true for countries like the former Soviet Union and East Germany, where a fair amount of literature dealing with resource and waste optimization is still accessible only in Russian or German. In Moscow, for example, a 'Department of Industrial Ecology' has been in operation for almost two decades at the Mendeleiev Institute of Chemical Technology (Zaitsev 1993; Ermolenko 1994; Melkonian 1994; Kirakossian and Sorger 1994).

In East Germany, during the 1960s, cybernetics and systems approach became increasingly part of the country's planned economy official thinking (Altmann *et al.* 1982; Busch *et al.* 1989), under the direct leadership of Walter Ulbricht, Seretary General from 1950 to 1971 of the Sozialistische Einheitspartei Deutschland (or SED, or East German Communist Party). As a result, systematic efforts were undertaken to ensure the best possible use of wastes and by-products, and specific laws were even passed for that purpose (Reidel and Donner 1977).

One could find many examples of efforts to reduce waste and close material loops from the early days of the Industrial Revolution, like the work of Peter Lund Simmonds (1814–1897) in England (Desrochers 2000). In fact, certain industrial sectors like dyes and petrochemicals largely developed from making use of waste and by-products (Talbot, 1920; Spitz 1988). Sometimes, attempts to reduce waste were done in a systematic way, as in the case of the Committee on Elimination of Waste in Industry of the Federated American Engineering Societies in the early 1920s (CEWI 1921; Hays 1959; Haber 1964). On November 19, 1920, Herbert Hoover was elected the first President of the Federated American Engineering Societies. Among his first acts, he named 17 engineers for a Committee on Elimination of Waste in Industry, which completed in five months a detailed analysis of waste in six branches of industry: building industry, men's clothing manufacturing, printing, metal trades, and textile manufacturing (eight years later, the same Herbert Hoover became the 31st President of the USA).

However, it is important to remember that industrial ecology offers a much broader perspective than just reducing or using waste. Industrial ecology aims at the integrated management of all resources (not only waste), within the conceptual framework of scientific ecology. Thus, strictly speaking, industrial ecology could not have been imagined prior to the emergence and progressive elaboration of the concept of ecosystem (Tansley 1935; Golley 1993).

Among all the earlier attempts, however, two deserve to be mentioned in some detail here: the Belgium ecosystem research, and the ground-breaking work carried out in Japan.

THE 'BELGIUM ECOSYSTEM'

In 1983, a collective work called 'L'Ecosystème Belgique. Essai d'écologie industrielle' was published in Brussels by the *Centre de recherche et d'information socio-politiques* (CRISP), an independent research center associated with progressive circles in Belgium (Billen *et al.* 1983). The book summarizes the thinking of half-a-dozen intellectuals linked to the left-wing socialist movement. Inspired by *The Limits to Growth* (Meadows *et al.* 1972), and especially by the 'Letter' of Sicco Mansholt (Common Market Commissioner), this small group sought to fill a gap that persisted in standard, including left-wing, economic thinking. The small group comprised six people from different fields (biologists, chemists, economists and so on), who accomplished this work outside of their everyday occupations. Their idea was to produce an overview of the Belgian economy on the basis of industrial production statistics, but to express these in terms of materials and energy flows rather than the traditional, abstract monetary units.

The basic principles of industrial ecology are clearly expressed in this work as follows:

> To include industrial activity in the field of an ecological analysis, you have to consider the relations of a factory with the factories producing the raw materials that it consumes, with the distribution channels it depends on to sell its products, with the consumers who use them . . . In sum, you have to define industrial society as an ecosystem made up of the whole of its means of production, and distribution and consumption networks, as well as the reserves of raw material and energy that it uses and the waste it produces . . . A description in terms of circulation of materials or energy produces a view of economic activity in its physical reality and shows how society manages its natural resources. (Billen *et al.*, 1983, pp. 19, 21)

The group studied six main streams from this angle: iron, glass, plastic, lead, wood and paper, and food produce. One of the main findings was the so-called 'disconnection' between two stages of a stream. This means that 'two sectors in the same stream, which could be complementary and develop in close interaction with each other, are oriented in quantitatively/qualitatively divergent directions' (ibid., p. 31). For instance, 80 per cent of the net output of steel in Belgium is intended for export owing to the opening of European borders. Under the authority of the European Community of Coal and Steel (ECCS), the Belgian steel industry thus developed rapidly, without any relationship with the development of the metal-production sector. The opening of outside markets encouraged an excessive growth of a heavy steel industry aimed mainly at the export market, to the detriment of its specializing in more elaborate technological products. As a result, the steel industry was completely disconnected from the metal-construction sector, an unlinking that has made the Belgian steel industry very dependent on exports for selling a rather commonplace product, as a consequence of which it is vulnerable to competition on the world market while providing an inadequate response to domestic needs.

Another very significant example is that of the unlinking of farming and breeding (ibid., p. 67). In the traditional pattern, there was a certain balance between farming and breeding in a mixed farming concern: the by-products and waste of mixed farming were used to feed the livestock. The animal density remained low, and animal excrements (liquid and solid manure) constituted the basis for soil amendments, sometimes supplemented with mineral fertilizer. The 'modernization' of agribusiness has destroyed this pattern. Livestock, which has become much more important, is fattened with industrial feed made out of imported raw materials.

Breeding has thus progressively cut itself off from farming activities as far as food resources are concerned. The same is true of animal excrement: the considerable mass of excrement can no longer be completely used up because it far surpasses the manuring capacity of the farmland. In both cases (breeding and farming), the by-products have outstripped their natural outlets, and have become waste, with disposal problems.

The authors reached the conclusion that the general features of the way the Belgian industrial system works (that is, opening, specialization and sectoral unlinking), attest to the internationalization of the Belgian economy, and result in three main forms of dysfunction (ibid., p. 89):

1. The economic opening of the Belgian system leads to the ecological opening of the materials cycles. Consumption residues, which could constitute a resource, are increasingly considered as waste, the disposal of which is a problem.
2. Operation of this economic system requires large energy expenditure. On this point, the analysis of the Brussels group particularly highlights the fact that the increase in primary energy comes less from the increase in end consumption than from a certain type of organization of the energy chain itself, as well as of the industrial system as a whole.
3. The structure of the circulation of materials in the industrial system generates pollution. For example, the present organization of the food chain causes the degradation of surface water.

The Belgian group also developed some interesting ideas on the subject of waste, by underscoring the fact that the notions of 'raw materials' and 'waste' only mean something from the point of view of a system where the circulation of materials is open. Contrary to the current assumption, in which the waste problem is seen as being due to an increase in production and consumption, 'our consumption of raw materials and our production of waste constitute a consequence of the structure of the circulation of raw materials in our industrial system'. As for the recycling of waste, we have to realize that 'the main difficulties are found not at the collection, or even at the sorting stage, but upstream of collection, that is, in the real possibilities of waste disposal in the current structure of our production system' (ibid., p. 91).

According to Francine Toussaint, main instigator of the project and a trade engineer currently working for the Brussels administration, the expression 'industrial ecology' seems to have come up on its own, spontaneously, without having been read or heard elsewhere.

Even though the work summarized the basic ideas of industrial ecology with remarkable clarity, its reception was extremely reserved. 'We really had the feeling that we were a voice preaching in the desert,' Toussaint recalls. Eventually, the group of friends branched off in different directions, each pursuing their career, and despite its interest and originality, the 'Belgium Ecosystem' was soon forgotten. However, the book had not gone totally unnoticed. For example, in Sweden at Lund University, Stefan Anderberg, one of the pioneers of regional material flow studies, refers to it in a 1989 publication (Anderberg *et al.* 1989). Ultimately, the interest in the work of the Belgian team was revived at the end of the 1990s, when one member of the group, Gilles Billen, a microbiologist, introduced the industrial ecology approach in a major environmental research program on the River Seine Basin (Billen 2000).

THE JAPANESE VIEW

Japan deserves particularly to be mentioned in the history of industrial ecology. In the late 1960s, the Ministry of International Trade and Industry (MITI), noting the high environmental cost of industrialization, commissioned one of its independent consulting agencies, the Industrial Structure Council, to do some prospective thinking. About 50 experts from a great variety of fields (industrialists, senior civil servants, representatives of consumer organizations and so on) then explored the possibilities of orienting the development of the Japanese economy toward activities that would be less dependent on the consumption of materials, and based more on information and knowledge. During the 1970 Industrial Structure Council session, the idea came up (without its being possible, apparently, to attribute it to a specific person) that it would be a good thing to consider economic activity in 'an ecological context'.

The final report of the Industrial Structure Council, called 'A Vision for the 1970s', was made public in May 1971. Complying with the recommendations of the report, the MITI immediately set up about 15 work groups. One of these, the Industry–Ecology Working Group, was specifically commissioned to further develop the idea of a reinterpretation of the industrial system in terms of scientific ecology.

The small group was coordinated by Chihiro Watanabe, an urban engineer, who was then in charge of environmental problems within a MITI agency, the Environmental Conservation Bureau (having occupied a variety of positions in MITI over 26 years. Watanabe is today a professor at the Tokyo Institute of Technology and adviser to the director of IIASA). With the assistance of several outside experts, the members of the Industry–Ecology Working Group began by conducting systematic research of the scientific literature, then consulted the best international specialists. It was in the course of a US tour, in March–April 1973, that Watanabe met one of the great figures of modern ecology, Eugene Odum, at Georgia State University, in Atlanta (who, nonetheless, did not appear to be particularly interested in the Japanese approach). After a year's work, in May 1972, the Industry–Ecology Working Group published its first report, a Japanese document of more than 300 pages, a summary of which is available in English (MITI 1972a, 1972b; Watanabe 1973).

According to Watanabe, the report was widely distributed within the MITI, as well as among industrial organizations and the media, where it was considered to be 'stimulating' but also still very 'philosophical'. A second, more concrete report, including case studies, was published a year later, in the spring of 1973. It is difficult to evaluate the exact legacy of the Industry–Ecology Working Group, but there is no doubt that its approach has contributed greatly to the design and implementation of many important MITI research programs on industrial technology. In April 1973, for instance, the secretariat of the minister in charge of MITI officially recommended that a new policy be developed on the basis of the ecology principle, with the accent on energy aspects.

In August 1973, two months before the first oil shock, MITI submitted a first budget request for the Sunshine Project. The project, which aimed to develop new energy technology (particularly in the area of renewable energy), was started in July 1974. A few months before the second oil shock, in 1978, MITI launched a supplementary program, the Moonlight Project, devoted to technology intended to increase energy efficiency. In 1980, MITI founded the New Energy Development Organization (NEDO), then in 1988 launched the Global Environmental Technology Program (MITI 1988a). The New Sunshine Program, devoted to advanced energy technology in view of, among other objectives, achieving an important reduction in greenhouse gases, was finally started in 1993. The New Sunshine Program is itself a component of a broader program, New Earth 21 (MITI 1988b).

Without falling into the usual stereotypes on Japan (long-term strategic vision, systemic approach and so on), we have to acknowledge that it is the only country where ideas on industrial ecology were ever taken seriously and put into practice on a large scale, even though they were already diffusely present in the USA and Europe. The consequences of this are not to be neglected, given that it is through technology developed in the context of an economy that has fully integrated ecological constraints that Japan intends to maintain its status as a great economic power (Richards and Fullerton 1994; Watanabe 1992a, 1993). A basic principle underlies this strategy: to replace material resources with technology. This is why technological dynamics is at the heart of Japanese thinking on industrial ecology (Watanabe *et al.* 1991; Watanabe 1995a, 1995b; Inoue 1992; Yoshikawa 1995; Akimoto 1995).

This approach, however, is not original per se: research on technological dynamics and industrialization as a historical phenomenon has been pursued in Europe and the USA for many years by a number of authors, such as Jesse Ausubel and Arnulf Grübler (Ausubel and Langford 1997; Lee and Nakicenovic 1990; Grübler 1990, 1991, 1996, 1998; Nakicenovic 1990). But, whereas this thinking has been incorporated in long-term and large-scale industrial strategies in Japan, it has been traditionally mainly academic in the West.

A NEW DEPARTURE WITH *SCIENTIFIC AMERICAN*

Every year in September, the popular scientific monthly *Scientific American* publishes an issue on a single topic. In September 1989, the special issue was on 'Managing Planet Earth', edited by William C. Clark (Harvard University), himself an influential member of the early industrial ecology 'invisible college' (Clark and Munn 1986).

The issue featured an article by Robert Frosch and Nicholas Gallopoulos, both then at General Motors, called 'Strategies for Manufacturing' (the original title proposed by the authors, 'Manufacturing – The Industrial Ecosystem View', was not accepted!) (Frosch and Gallopoulos 1989).

In their article, the two authors offered the idea that it should be possible to develop industrial production methods that would have considerably less impact on the environment. This hypothesis led them to introduce the notion of the industrial ecosystem. Projections regarding resources and population trends

> lead to the recognition that the traditional model of industrial activity – in which individual manufacturing processes take in raw materials and generate products to be sold plus waste to be disposed of – should be transformed into a more integrated model: an industrial ecosystem. (. . .) The industrial ecosystem would function as an analogue of biological ecosystems. (Plants synthesize nutrients that feed herbivores, which in turn feed a chain of carnivores whose wastes and bodies eventually feed further generations of plants.) An ideal industrial ecosystem may never be attained in practice, but both manufacturers and consumers must change their habits to approach it more closely if the industrialized world is to maintain its standard of living – and the developing nations are to raise theirs to a similar level – without adversely affecting the environment. (Frosch and Gallopoulos 1989, p. 106)

However, as Frosch indicated during his lecture, 'Towards an Industrial Ecology', presented before the United Kingdom Fellowship of Engineering in 1990: 'The analogy between the industrial ecosystem concept and the biological ecosystem is not perfect, but much could be gained if the industrial system were to mimic the best features of the biological analogy' (Frosch and Gallopoulos 1992, p. 272).

On the occasion of the first symposium on industrial ecology, which took place in Washington in May 1991 under the authority of the National Academy of Science and chaired by Kumar Patel of Bell Labs (Patel 1992), Frosch pointed out that the idea had been around for a long time:

> The idea of Industrial Ecology has been evolving for several decades. For me the idea began in Nairobi with discussions at the United Nations Environment Program (UNEP), where we were concerned with problems of waste, with the value of materials, and with the control of pollution. At the same time, we were discussing the natural world and the nature of biological and ecological systems. There was a natural ferment of thinking about the human world, its industries, and its waste products and problems and about the coupling of the human world with the rest of the natural world. (Frosch 1992)

In contrast to preceding attempts, Frosch and Gallopoulos's article sparked off strong interest. There are many reasons for this: the prestige of the *Scientific American*, Frosch's reputation in governmental, engineering and business circles, the weight carried by the authors because of their affiliation with General Motors, and the general context, which had become favorable to environment issues, with, among other features, discussions around the Brundtland Commission report on sustainable development. The article manifestly played a catalytic role, as if it had crystallized a latent intuition in many people, especially in circles associated with industrial production, who were increasingly seeking new strategies to adopt with regard to the environment.

Although the ideas presented in Frosch and Gallopoulos's article were not, strictly speaking, original, the *Scientific American* article can be seen as the source of the current development of industrial ecology. In Washington, the National Academy of Engineering (NAE) had shortly before launched the Technology and the Environment Program, organizing symposia and publishing their reports. The first of these, published in 1989, 'Technology and the Environment', already contains many of the ideas that evolved in the direction of industrial ecology, and was followed by a number of volumes on industrial ecology (Ausubel and Sladovich 1989; Allenby and Richards 1994; Richards and Frosch; 1994; Schulze 1996; Richards 1997; Richards and Pearson 1998). Braden Allenby, an AT&T executive who spent in 1991/1992 a one-year fellowship with the NAE Technology and the Environment Program, presented the first doctoral dissertation on industrial ecology in 1992 (Allenby 1992a), and then published in 1995 the first textbook on industrial ecology, with Thomas Graedel (at the time also at AT&T, and who became, in 1997, the first professor of industrial ecology in the USA, at Yale University) (Graedel and Allenby 1995; Graedel 1996).

REACHING THE BUSINESS AND ACADEMIC COMMUNITIES

Ideas on industrial ecology were also disseminated among business circles on the basis of the *Scientific American* article, but indirectly. Hardin Tibbs, a British consultant who was working in Boston in 1989 for the company Arthur D. Little, says that reading Frosch and

Gallopoulos's article inspired him to write a 20-page brochure called 'Industrial Ecology: A New Environmental Agenda for Industry'. Arthur D. Little published the text in 1991. It was published again in 1993 by Global Business Network, a consulting company based near San Francisco joined by Hardin Tibbs, which develops prospective scenarios for its member companies (Tibbs 1993).

In substance, Tibbs' brochure basically reproduces the ideas contained in the Frosch and Gallopoulos article, but Tibbs' decisive contribution was to translate them into the language and rhetoric of the business world, and to present them in a very summarized form in a document just a few pages long, stamped first with the Arthur D. Little label, then with that of the Global Business Network.

The Tibbs brochure was quickly sold out, then thousands of Xeroxed copies of it were circulated, spreading Frosch and Gallopoulos's ideas throughout the business world. Other authors, also inspired by the Frosch and Gallopoulos article, began to write papers disseminating the idea in various academic and business circles.

At the same time, a number of specific research themes started to be discussed under the umbrella of the industrial ecology concept: mainly industrial symbiosis (Gertler 1995; Gertler and Ehrenfeld 1996; Ehrenfeld and Gertler 1997), eco-industrial parks and eco-industrial networks (Schwarz 1995; Lowe *et al.* 1996; Lowe and Warren 1996; Côté *et al.* 1994; Weitz and Martin 1995). Some of these themes had already been investigated for decades before being reinterpreted within the industrial ecology conceptual framework, such as resource availability and intensity of use (Malenbaum 1978; Fischman 1980; Humphreys 1982; Humphreys and Briggs 1983; Auty 1985), resource productivity (Schmidt-Bleek 1993a, 1993b, 1993c, 1994a; Weizsäcker *et al.* 1997), transmaterialization (Waddell and Labys 1988), dematerialization (Herman *et al.* 1989; Bernardini and Galli 1993; Schmidt-Bleek 1994b; Welfens 1993; Wernick 1994; Wernick *et al.* 1996; Kanoh 1992; Socolow 1994), decarbonization (Ausubel 1996b; Ausubel and Marchetti 1996; Nakicenovic 1996; Socolow 1997), and the service or functionality economy (Stahel and Reday-Mulvey 1981; Stahel and Jackson 1993; Giarini and Stahel 1993). Finally, in 1997, eight years after the seminal paper of Frosch and Gallopoulos, the first issue of the *Journal of Industrial Ecology* was published (owned by Yale University and published by The MIT Press). The start of this journal can be seen as an official recognition by the academic community of the 'new' field of industrial ecology, which is now being pursued with unprecedented vigor.

4. Industrial ecology and cleaner production

Tim Jackson

Clean (or cleaner) production is an approach to environmental management which aims to encourage new processes, products and services which are cleaner and more resource-efficient. It emphasizes a preventive approach to environmental management taking into account impacts over the whole life cycle of products and services.

There are clear conceptual resonances between industrial ecology and cleaner production. Both are motivated by concerns about the increasing environmental impacts of industrial economic systems. They emerged at more or less the same time (the late 1980s to mid-1990s) in the evolution of environmental management. Both have spawned their own journals and their own literature. A brief survey of this literature reveals strong intellectual overlaps between the two models. For example, the *Journal of Cleaner Production* (published by Elsevier Science) advertises its scope as including the following concepts:

- pollution prevention,
- source reduction,
- industrial ecology,
- life cycle assessment,
- waste minimization,
- sustainable development.

Thus cleaner production claims to include industrial ecology within its remit, and has on one occasion devoted a special issue of the journal to industrial ecology (Ashford and Côté 1997). At the same time, the *Journal of Industrial Ecology* (published by The MIT Press) 'focuses on the potential role of industry in reducing environmental burdens throughout the product life cycle from the extraction of raw materials, to the production of goods, to the use of those goods, to the management of the resulting wastes'. Without explicitly using the term, the journal's list of topics includes much of the ground covered by cleaner production.

In spite of these similarities and overlaps, the two concepts emerged in slightly different ways from slightly different places, and there are, at least on some interpretations, discernible differences in approach which flow from these historical idiosyncrasies. This chapter sketches briefly the history of the concept of cleaner production and its integration into a network of activities coordinated by the United Nations Environment Programme (UNEP). It next sets out some of the underlying principles of cleaner production and describes how these are translated into operational strategies. Finally, it discusses key similarities and differences between cleaner production and industrial ecology.

A BRIEF HISTORY OF CLEANER PRODUCTION

The concept of industrial ecology was introduced in the 1990s in Japan (see Chapter 20). It re-emerged primarily in the USA in response to an article by Frosch and Gallopoulos (1989) and later through initiatives such as the 1992 Global Change Institute (Socolow *et al.* 1994) and the work of Graedel and Allenby (1995). However, it drew heavily on the earlier concept of industrial metabolism (see Chapter 2) developed in particular by Ayres *et al.* (1989).

By contrast, the terminology of cleaner production emerged primarily in Europe and through the initiatives of the Industry and Environment office of the United Nations Environment Programme (UNEP). However, it also drew heavily from earlier concepts, in particular from the terminology of 'clean technology' articulated for example by the Organization for Economic Cooperation and Development (OECD) who defined clean technologies as 'any technical measures . . . to reduce, or even eliminate at source, the production of any nuisance, pollution or waste, and to help save raw materials, natural resources and energy' (OECD 1987).

The term 'clean production' itself was coined in May 1989 at a meeting in Paris convened to advise the UNEP Industry and Environment office on the development of a new global information network on low and non-waste technologies. The word 'technology' was replaced by the word 'production' in this meeting, because the committee felt that the earlier term suffered from a critical lack of emphasis on the complex of social, economic and ecological factors which needed to be addressed if progress was to be made towards sustainable development (Baas *et al.* 1990). The meeting defined clean production as: 'the conceptual and procedural approach to production that demands that all phases of the life cycle of a product or a process should be addressed with the objective of prevention or minimization of short and long-term risks to humans and to the environment' (ibid., p. 19).

The terminology was later amended to 'cleaner production' on the recognition that no process or product chain could be expected to be entirely without environmental impact or potential adverse health effects. Cleaner production was supposed to indicate a progressive program of improvements in the environmental performance of industrial processes and product systems.

UNEP's Cleaner Production Program was formally launched in September 1990 at the first biennial 'high-level' Seminar on Cleaner Production held in Canterbury in the UK. Six high-level seminars have been held in total since that time, the last one being in Montreal in October 2000. By the end of the decade, the Cleaner Production network sponsored in part by the UNEP initiative comprised more than 140 cleaner production centers in over 40 countries (Aloisi de Lardarel 1998). The Industry and Environment office had also organized training workshops, supported a number of regional round table workshops, and developed an extensive database of cleaner production case studies known as the International Cleaner Production Information Clearinghouse (ICPIC) available on the web. The 1998 meeting in Seoul formalized an *International Declaration on Cleaner Production*. Recognizing that achieving sustainable development is a collective responsibility, the declaration calls for action to protect the global environment to include the adoption of improved sustainable production and consumption practices, and to prioritize cleaner production and other preventive environmental strategies such as eco-efficiency, green productivity and pollution prevention. As of the beginning of 2000, there were 172 high-level signatories of the Declaration (UNEP 2000).

PRINCIPLES OF CLEANER PRODUCTION

The most recent formal definition of the concept of cleaner production is the one contained in the cleaner production declaration which defines cleaner production as 'the continuous application of an integrated, preventive environmental strategy applied to processes, products and services in pursuit of economic, social, health, safety and environmental benefits'. Both this more recent definition and the original definition cited rest basically on three main 'guiding principles' which distinguish cleaner production from earlier environmental management strategies. Jackson (1993) identified these guiding principles as precaution, prevention and integration.

First of all, the lessons of the precautionary principle (Raffensberger and Tickner 1999; Sand 2000) are clearly relevant in structuring a new approach to environmental protection. This principle emerged as an important factor in environmental policy at around the same time as cleaner production emerged as a new environmental management paradigm. The earliest formulation of the principle can be traced back to the first international Conference on the Protection of the North Sea in 1984 (Dethlefsen *et al.* 1993). The second conference, in 1987, formalized acceptance of the principle by agreeing to 'reduce polluting emissions' of particular kinds of substances 'especially where there is reason to assume that certain damage or harmful effects . . . are likely to be caused by such substances' (North Sea Ministers 1987). The fundamental import of the principle is to take action to mitigate potential causes of environmental pollution in advance of conclusive scientific evidence about actual effects. Though originally formulated in terms of a specific class of substances – namely those that are persistent, toxic and bioaccumulable – subsequent applications of and attempts to explicate the principle have stressed that the domain of precaution could potentially be applied to all anthropogenic emissions. As such, the principle enshrines a call to reduce the material outputs from all industrial systems: in effect therefore to engage in cleaner production.

The principle of prevention provides perhaps the most fundamental distinction between the concept of cleaner production and earlier environmental protection strategies (Hirschhorn and Oldenburg 1991; Hirschhorn *et al.* 1993). The idea of a preventive approach to problem solving can be illustrated by reference to preventive health care. Curative medicine attempts to correct imbalances and diseases in the organism through surgery or through treatment with drugs. Preventive medicine seeks to prevent illness itself by promoting health in the patient, and increasing his or her natural resistance to disease. But preventive medicine, to be successful, must act upstream, as it were, in advance of the onset of disease. Once the illness has set in, the organism is already out of balance. Curative medicine can of course still 'prevent' a sick patient from dying, and often aids recovery. But it is generally more expensive and often more difficult than ensuring that the patient stays healthy to start with.

Preventive environmental management also requires actions to be taken upstream, before environmental impacts occur. This is in contrast to more traditional environmental management strategies which by focusing on environmental endpoints tend to clean up pollution, as it were, after the fact. Such clean-up strategies can sometimes 'prevent' environmental emissions from affecting human health, and for this reason remain important within environmental management. But they are expensive ways of dealing with anthropogenic impacts on the environment, and generally fail to address the root causes of pollution. Preventive environmental management also distinguishes itself from end-of-pipe environmental management which attempts to 'prevent' the emission of specific pollutants into a particular environmental medium by placing some kind of filter or treatment between the emission and the environment. Again, the logic of prevention is to seek intervention at an earlier stage of the process in such a way that the polluting emission does not arise in the first place.

There is a sense in which the prevention is thus a directional strategy: it looks as far as possible upstream in a network of causes and effects; it attempts to identify those elements within the causal network which lead to a particular problem; and it then takes action at the source to avoid the problem. The preventive approach recognizes the demand for products and services as the prime mover in the impact of anthropogenic systems on the environment. In particular, therefore, the preventive nature of clean production entails the need to 'reconsider product design, consumer demand, patterns of material consumption, and indeed the entire basis of economic activity' (Jackson 1993).

Finally, cleaner production attempts to formulate an integrated approach to environmental protection. Traditional end-of-pipe approaches have tended to concentrate on specific environmental media: air, water or land. One of the failures of earlier management approaches was to reduce specific environmental emissions at the expense of emissions into different media. Cleaner production attempts to avoid this problem by concentrating on all material flows, rather than selected ones. Furthermore, as the definitions point out, cleaner production demands that attention be paid to emissions over the whole life cycle of the product or service from raw material extraction, through conversion and production, distribution, utilization or consumption, re-use or recycling, and to ultimate disposal.

OPERATIONAL PATHWAYS TO CLEANER PRODUCTION

The conversion of these guiding principles into an operational strategy is of course a complex, multifaceted task. It is also highly dependent on sector-specific and application-specific parameters. Nonetheless it is possible to identify some specific types of action which flow from the principles of cleaner production. In particular it is possible to identify two main 'operational pathways' for clean production (Jackson 1993, 1996). In the first place, the environmental impacts of processes, product cycles and economic activities are minimized by reducing the material flow through these processes, cycles and activities. If this reduction in material flow is to occur without loss of service, then this strategy implies the pursuit of *efficiency improvements* in the system. Efficiency improvement is thus the first operational pathway of clean production. This pathway has been the subject of several similar strategic approaches to environmental improvement put forward, for example, under the concept of Materials Intensity Per unit of Service (MIPS) by Schmidt-Bleek (1993a), and Factor 4 or Factor 10 efficiency improvements by von Weizsäcker *et al.* (1996) and others. The second operational pathway is that of *substitution*: specifically, the substitution of non-hazardous or less hazardous materials for hazardous materials in processes and products.

These two operational pathways mean different things at different system levels. In terms of production processes, efficiency improvements might range from simple good housekeeping actions and better materials handling to redesigning process technologies in order to close material loops within the process. These kinds of techniques developed on the back of efficiency-oriented pollution prevention programs such as those promoted by companies like 3M, Dow Chemical, Dupont and Chevron in the 1970s and 1980s. Slogans such as 3M's 'pollution prevention pays' (the 3P program) and the Chevron Corporation's 'save money and reduce toxics' (the SMART program) highlighted what had been a fundamental truth of corporate economics ever since the industrial revolution (Jackson 1996): reducing material input costs while maintaining output revenues increases corporate profitability; or, in other words, improving material efficiency is cost-effective. For example, a New Jersey facility of the oil company EXXON achieved a 90 per cent reduction in evaporative losses from its chemical storage tanks by simply installing 'floating roofs' on those tanks containing the most volatile chemicals (Dorfman *et al.* 1992). As well as reducing the emission of volatile organic compounds at the source, this action saved the company \$200 000 per year.

The second operational pathway – replacement of toxic or hazardous input materials – is less readily driven by economic goals, and therefore, not surprisingly, less common. A study of 29 organic chemical companies through the 1980s found that only around 10 per cent of the reported actions involved substitution (Dorfman *et al.* 1993). Those actions that did involve substitution were usually driven primarily by regulatory pressures to reduce hazardous wastewater discharges or to phase out the use of particular chemicals, a trend which appears to have intensified subsequently (Verschoor and Reijnders 2000). Nonetheless, such regulatory pressures can also provoke economic savings. For example, a Monsanto plant which modified its product to substitute one kind of formaldehyde resin for another simultaneously reduced its hazardous waste generation by 89 per cent, saving the company around \$60 000 annually.

In terms of product and service system levels, it still makes sense to talk of efficiency improvement – providing the same level of service but with lower material throughput – and substitution – replacing specific harmful products or activities with less harmful ones. For example, in product systems, the efficiency pathway corresponds to a set of strategies which aim to re-use, recondition and recycle products and their material constituents. A variety of companies have experimented with changes in corporate structure which allow revenue to be gained from leasing durable products, rather than selling them, and taking on the responsibility of repair, reconditioning and ultimately 'take back' of the material products after use (Stahel and Jackson 1993). This kind of trend is now being reinforced by European legislation on take back in the electronics and automotive industries, for example.

This set of strategies for efficiency improvement at the level of product and service systems comes perhaps closest to the strategic thrust of industrial ecology. However, it is perhaps worth noting that efficiency gains at the different system levels are not necessarily correlated. For instance, it is clearly in the commercial interests of industry to pursue material efficiency at the process level. But it is also in the commercial interest of industry to sell as many material products as possible; efficiency at the level of the product loop involves actors other than those involved at the process level, and requires efforts other than those involved in good housekeeping or improvements in process technology. Ironically, process efficiency measures could sometimes even end up reducing material efficiency at the product level. Furthermore, it is clear that even improved material efficiency at the level of an individual product or service system does not necessarily ensure macroeconomic material efficiency, much less overall reductions in material throughput.

Thus cleaner production may be seen to entail a rather broad set of actions which certainly extend beyond technological measures to reduce waste emissions from industrial production processes. A number of different authors have attempted to define this wider set of actions. Both Jackson (1993) and Misra (1995), for example, are clear that, if cleaner production is to provide an appropriate environmental management strategy, it must pay attention to the question of consumption as well as that of production. Certainly, as Ayres (1993b) makes clear, cleaner production at the process technology level is not enough. In fact, as he points out, 'if every factory in the world shifted to the cleanest available (or even the cleanest plausible) technology, the larger environmental crisis would be, at best, deferred by a few years'. This same message is reflected in a pamphlet on clean production published by the environmental campaign lobby Greenpeace (1992), who were early champions of the concept. The pamphlet defines 'fifteen steps to clean production' as follows:

- consume consciously and consume less,
- conserve resources and use only renewables,
- establish community decision making,
- require public access to information,
- ensure worker protection,
- convert to chemical-free food and textiles,
- mandate clean production audits,
- eliminate toxic emissions and discharges,
- stop toxic waste disposal,
- phase out toxic chemical production,
- ban hazardous technology and waste trade,
- prohibit toxic waste recycling,
- prosecute corporate criminals,
- be active in your community,
- support Greenpeace.

Though some of these actions (for instance the last!) might appear to have more to do with environmental banner waving than with defining an operational environmental management strategy, their breadth indicates the promise which proponents of clean production hold out for the concept. It is seen, in its broadest terms, as nothing less than a wholesale strategy for the pursuit of sustainable development.

CLEANER PRODUCTION V. INDUSTRIAL ECOLOGY: DISCUSSION AND CONCLUSIONS

One of the similarities between cleaner production and industrial ecology is that both concepts have been dogged by definitional problems and even now admit of multiple interpretations.

Much of the early emphasis of the UNEP cleaner production program was on process technology. Under this interpretation of the term, cleaner production owed much to the earlier concept of pollution prevention (USOTA 1987; Hirschhorn and Oldenburg 1991). There remains a tendency for the UNEP cleaner production program to focus its efforts on process technology improvements rather than the more intractable problems associated with consumption patterns, or even product take back and recycling. The UNEP program's network of national cleaner production centers focus mainly on providing information on the potential for pollution prevention and waste minimization opportunities in small and medium-sized enterprises. Many of the articles published in the *Journal of Cleaner Production* cover the same sorts of topics. At the same time, as noted above, there is an agreement that these kinds of actions do not exhaust the remit of cleaner production, and far broader interpretations of the concept exist, as the previous section made clear.

Equally, industrial ecology is interpreted with varying degrees of breadth or specificity. Under some interpretations it is simply a way of focusing attention on the use or re-use of the wastes generated by one industrial process as material inputs to sister processes (for example, Lowe 1997). Under broader interpretations industrial ecology is nothing less than 'an integrated systems-perspective examination of industry and environment [which] conceptualizes the industrial system as a producer of both products and wastes and examines the relationships between producers, consumers, other entities and the natural world' (Sagar and Frosch 1997). Though couched in terms of 'examining the relationship', the often implicit goal of industrial ecology under this broader interpretation is to reduce the impact of the industrial system on the environment – or, more broadly still, to pursue sustainable development. Thus under the broader interpretations the concepts of cleaner production and industrial ecology tend to approach each other closely.

Differences are more obviously apparent under the narrower interpretations of the concept. In particular, as Oldenburg and Geiser (1997) point out, both the scope and the locus of actions are different in each case. While both cleaner production and industrial ecology focus on the concept of material efficiency, cleaner production (like pollution prevention) ascribes an equally important role to hazard reduction through substitution, suggesting the reduction or complete phase-out of use of certain priority toxics. Furthermore, cleaner production actions (in the narrower sense of pollution prevention) are assumed to be carried out more or less autonomously by and within individual firms. Industrial ecology, on the other hand, relies more heavily on the relationship between firms, and therefore requires cooperative networks of actors engaged at a different functional level than those in process-focused cleaner production activities.

Oldenburg and Geiser also argue that pollution prevention occupies a more specific role within a better defined regulatory structure than does industrial ecology. Clearly, there is an argument to suggest that this observation is truer of the USA, where the concept of pollution prevention is perhaps better enshrined in federal and state legislative initiatives than it has been in other countries. Nonetheless, it is certainly true that industrial ecology is not currently driven by regulatory initiatives. Rather it operates on the basis of industrial cooperation, driven mainly by the economic advantages of re-using waste resources.

In a sense each of these differences between the two concepts highlights potential drawbacks within the individual concepts. For example, it is clear that a single-minded focus on materials efficiency in the large could potentially overlook the priority hazards associated with the use of particularly toxic materials. Thus industrial ecology can learn from cleaner production the importance of the substitution pathway. On the other hand, it is clear that relying entirely on autonomous pollution prevention strategies within individual firms is unlikely to lead to material efficiency, or indeed dematerialization, at the wide system or macroeconomic level. Thus cleaner production can learn from industrial ecology, as Pauli (1997) points out, the importance of cooperative relationships between individual firms in the drive for sustainable development.

When it comes to comparing and contrasting the broader interpretations, it is far harder to distinguish between cleaner production and industrial ecology. Each claims to provide an operational strategy for achieving sustainable development, and tends to expand its own definition to include whatever might be necessary to achieve these ends. There is a sense therefore in which cleaner production and industrial ecology can be regarded as rivals for the same intellectual territory. Which of the two concepts is ultimately successful in occupying that territory is probably less important than that the lessons from developing and operationalizing the individual concepts be directed towards what appears to be the common end of both.

5. On industrial ecosystems

Robert U. Ayres*

Industrial ecosystems, designed 'from scratch' to imitate nature by utilizing the waste products of each component firm as raw materials (or 'food') for another, are an attractive theoretical idea, but as yet mostly at the proposal stage. It is important to stress that process changes to take advantage of returns to closing the materials cycle are very definitely *not* another version of 'end-of-pipe' treatment of wastes. Is this an idea whose time has come?

This chapter examines a number of such proposals and considers the prerequisites for success. It appears that there are several. First, a fairly large scale of operation is required. This means that at least one first-tier exporter must be present to achieve the necessary scale. Second, at least one other major firm (or industrial sector) must be present locally to utilize the major waste of the exporter, after conversion to useful form. Third, one or more specialized 'satellite' firms will be required to convert the wastes of the first-tier exporter into useful raw materials for the consumer, and to convert the latter's wastes into marketable commodities, secondary inputs to other local firms, or final wastes for disposal. A final condition, of great importance (and difficult to achieve in practice) is that a reliable mechanism be established to ensure close and long-term cooperation – that is, information sharing – at the technical level among the participating firms. The guarantor of this cooperation must be either the first-tier exporter itself, a major bank, a major marketing organization or a public agency. The detailed mechanisms by which it can be achieved in practice remain to be worked out.

RETURNS TO SCALE AND SCOPE

The notion of returns to scale – and the related notion of division of labor – are among the oldest and most familiar insights in economics, going back at least to Adam Smith. It is not necessary to expound them in detail again. However, it is helpful (for what follows) to remind the reader that the classic 'supply–demand' intersection presupposes a rising supply curve, for the economy as a whole, or for a particular good or service. This implies that the *marginal* cost of production (of the good or service) rises monotonically with output. A rising marginal cost curve reflects short-term rigidities, namely, a fixed workforce, a fixed physical capital and a fixed technology.

At first sight this equilibrium picture seems incompatible with economies of scale, a point that properly bothers many thoughtful first-year economics students. However,

* This chapter is reprinted in full from Chapter 15 of Robert U. Ayres and Leslie W. Ayres (1996), *Industrial Ecology: Towards Closing the Materials Cycle*, Cheltenham, UK and Brookfield, US: Edward Elgar. With permission of the publishers.

economies of scale become meaningful when we relax the assumption of a fixed set of factors of production at a static point in time. Absent the (implied) condition of short-term rigidity, it is easier to see that alternative technologies of production can, and do, exist, *in principle*, at different scales of operation. They are likely to differ in many ways, especially in terms of capital/labor ratio. At larger scales of operation, and especially with longer production runs, more operations can be carried out by machines or other equipment than at small scales. The physical reasons why this is true range from the obvious (longer runs = less set-up time) to the arcane (decreasing surface/volume ratios) and need not be considered further here. The important consequence is that the number of workers (or man-hours) per unit output tends to decline with increasing scale, *ceteris paribus*.

This fact, in turn, has had a major impact on economic growth in the past. Passing from a static to a dynamic framework, consider what happens when existing productive capacity in an industry is augmented by a new and more efficient plant. The new plant can produce more cheaply. Potential supply in the industry has increased. Demand will absorb the larger supply only at a lower price. If competing suppliers cut prices, demand will continue to rise. This will stimulate further investment in supply, and so on. The cycle of price cuts (permitted by economies of scale) leading to increasing demand (thanks to price elasticity of demand) has been called the 'Salter cycle'.

To maximize economies of scale, manufacturers in the late 19th and early 20th centuries adopted a strategy of product standardization and 'mass production'. These were basically US innovations. Successful exemplars ranged from Waltham watches, Colt 45 revolvers, Remington rifles, Yale locks and Singer sewing machines, to Ford's successful 'Model T' (which was produced continuously from 1908 to 1926). At that time, the keys to success in manufacturing were product standardization, division of labor and volume (Ayres 1991b). Frederick Taylor incorporated these elements, together with some others, into a formal theory of 'scientific management' which strongly influenced (and was influenced by) Henry Ford.

The benefits of scale are diluted in the case of mechanical or electrical products which are evolving and improving over time. Mechanical automation requires large investments in specialized machinery and equipment, which must be depreciated. This tends to discourage technological change, since new models require new and costly production lines to be designed and custom-built (see, for example, Abernathy 1978). The benefits of scale are most obvious (and easiest to analyze econometrically) in the case of homogeneous commodities such as steel, petrochemicals or electric power. Economies of scale tend to encourage industrial gigantism, and oligopolies, at the expense of competition. Economists have argued the relative benefits to consumers of scale economies v. competition in regulated utilities, such as telecommunications, electric power, water and gas distribution, railroads or airlines (see, for example, Christensen and Greene 1976). Evidently, oligopolistic pricing inhibits growth, but it appears that economies of scale were still an important engine of economic growth even for the USA, and much more so for Europe and Japan, in the post-war decades (see, for example, Denison 1962, 1974, 1979). Scale economies were perhaps the *only* significant growth factor for the Soviet Union and Eastern Europe during that period.

Recently, there have been indications that economies of scale are no longer as important as they once were. Markets for many standardized products have become saturated, at least in the West. Quantity of supply (of final goods and services) is less and less important relative to quality and variety. From the classical Taylorist–Fordist point of view, variety (diversity) of output is incompatible with maximum efficiency. Yet there are other, hitherto neglected, dimensions of production technology and other strategies for cost reduction that are more appropriate for meeting demand in markets where diversity – even ‘customization’ – is inherently valuable. Newer strategies maximizing ‘returns to scope’ (or ‘economies of scope’) have become increasingly important in recent years, thanks to the introduction of new computer-based technologies in manufacturing. In brief, the idea is that a manufacturer who can produce a large number of different products *efficiently* from a small number of flexible workers or programmable machines will be more able to meet variable demand than a manufacturer with inflexible machines geared to a single standardized product. It appears that advanced forms of computer-controlled production, linked with computer-assisted design and engineering – known as computer-integrated manufacturing, or CIM – offer a feasible path away from traditional mass production (see Goldhar and Jelinek 1983, 1985; also Ayres 1991a).

Another hitherto neglected dimension of strategy (with a few exceptions) is to maximize *systems integration*. We consider this strategy in more detail next.

RETURNS TO SYSTEMS INTEGRATION

Having emphasized that viable production strategies today no longer depend exclusively on standardization and economies of scale, one can look more systematically for other sources of competitive advantage. In particular, we wish to consider ‘returns to integration’, or ‘returns to internalization’ (that is to say, ‘closure’) of the materials cycle.

The classical illustration of this strategy was the Chicago meat packers who prided themselves on recovering and finding markets for ‘everything but the squeal’ of the slaughtered animals. (See, for example, Siegfried Giedion’s *Mechanization Takes Command*, Part IV, 1948, pp. 213–40). The link between scale and integration is obvious: only a large-scale operator could invest in the various specialized facilities needed to produce various meat products from steak to sausage, lard (some of which was saponified to produce soap), pet-food, bone-meal, blood-meal, gelatin (from hooves) and even hormones from animal parts. Pig bristles became shaving brushes and hairbrushes, while the hides were tanned to make leather.

Coke, used in blast furnaces, offers another historical example. The earliest ‘beehive’ coke ovens were terrible polluters. However, the ‘by-product’ coking process, first introduced by Koppers, in Germany, changed this situation significantly by capturing both the combustible gas and other by-products of the coking operation. (Even the most modern coke ovens are not regarded as desirable neighbors, since there are still non-negligible emissions from leaks, dust and especially from the quench-water used to cool the red-hot coke.) High-quality coke oven gas became available near the Ruhr steelworks in the late 19th century. Its availability encouraged a local inventor, Nicolaus Otto, to commercialize a new type of compact ‘internal combustion’ engine – to replace the bulky steam engine (and its associated furnace, boiler and condenser) – to supply power for small factories. The Otto-cycle gas engines were quickly adapted to liquid fuels, higher speeds and smaller sizes by one of Otto’s associates, Gottlieb Daimler. The Daimler engine, in turn, made possible the motor vehicle and the airplane.

By-product coke ovens also produced coal tar. Coal tar was the primary source of a number of important chemicals, including benzene, toluene and xylene, as well as aniline. Aniline was the raw material for most synthetic organic dyestuffs, the first important product of the German chemical industry. Coke ovens were also the source of most industrial ammonia, which was the raw material for both fertilizers (usually ammonium sulfate) and nitric acid for manufacturing explosives such as nitroglycerine.

The modern petrochemical industry is the best current example of systems integration: it begins with a relatively heterogeneous raw material (petroleum), which consists of a mixture of literally thousands of different hydrocarbons. The first petroleum refiners of the 19th century produced only kerosine ('illuminating oil') for lighting and tar for road surfaces and roofing materials. The lightest fractions were lost or flared; even natural gasoline had few uses (except as a solvent for paint) until the liquid-fueled internal combustion engine appeared on the scene in the 1890s. By the second decade of the 20th century the market for petroleum products had become mainly a market for automotive fuels; after 1920 this market expanded so rapidly as to create a need for 'cracking' heavier fractions and, later, recombining lighter fractions (by alkylation) to produce more and more gasoline. Heavier oils found uses in diesel engines, as lubricants, as fuel for heating homes and buildings, and as fuel for industrial boilers and electric power plants. Meanwhile, a great deal of natural gas was found in association with petroleum deposits, and a beginning was made on the long process of capturing, processing and utilizing this new resource.

By the 1930s, by-products of petroleum and natural gas process engineering began to find other chemical uses. In particular, ammonia and methanol were derived from methane, from natural gas. Then ethylene (produced, at first, by pyrolysis of ethane separated from natural gas) became cheap, as did hydrogen. This encouraged the development of a family of synthetic polymers, starting with polyethylene and followed by polyvinyl chloride. Natural gas liquids and light fractions of petroleum refining also became the basis of most synthetic rubber production (via butadiene). Propylene, from propane, is now second only to ethylene as a chemical feedstock. Another whole family of chemicals was created from benzene, also a by-product of petroleum refining. Phenol, the basis of polystyrene and phenolic resins, is a benzene derivative. Finally, the refining process has become a major source of sulfur.

Most of the organic chemicals and synthetic materials produced today are derived from one of these few basic feedstocks. A very sophisticated technology for converting a few simple hydrocarbon molecular structures into others of greater utility has arisen. Light fractions become chemical feedstocks. The heavier fractions, including asphalt, are the least valuable (per unit mass), but some products of the heavy fractions – like lubricants and petroleum coke – are very valuable indeed. It is fair to say that, today, there are scarcely any wastes from a petroleum refinery. There is a continuing trend towards adding value to every fraction of the raw material. While most petroleum products are still used for fuel, the fuel share is actually declining and the share of fuel for stationary power plants and space heating is declining quite fast. It is virtually certain that these 'low value' uses of petroleum will be displaced in a few decades by higher value uses without any intervention by governments.

To be sure there are other chemical families where by-products have been much harder to utilize. One example is biomass. Cellulose and cellulosic chemicals (such as rayon) are derived from wood, but about half of the total mass of the harvested roundwood – lignins – is still wasted or burned to make process steam. There are a few chemical uses of lignins, but no more than a few per cent of the available resource is used productively except as low-grade fuel for use within the pulp/paper plants.

The idea of converting wastes into useful products via systems integration has recently become a popular theme among environmentalists. The 3M company introduced a formal program, beginning in 1975, with the catchy title: 'Pollution Prevention Pays' or PPP. The Dow Chemical Co. has its own version, namely 'waste reduction always pays' or WRAP. Others have followed suit. These acronyms are not merely 'awareness raisers' to attract the attention of managers and staff; they also contain an element of generalizable truth. Unquestionably, it is socially and environmentally desirable to convert waste products into salable by-products, even though the economics may be unfavorable at a given moment of time. However, the economic feasibility of converting wastes into useful products often depends on two factors: (1) the scale of the waste-to-by-product conversion process and (2) the scale of demand (that is, the size of the *local* market).

For example, low-grade sulfuric acid is not worth transporting but it can be valuable if there is a local use for it. Thus sulfuric acid recovered from copper smelting operations is now routinely used to leach acid-soluble oxide or chalcocite copper ores; the leachate is then collected and processed by solvent extraction (SX) technology and electrolytically reduced by the so-called 'electro-winning' (EW) process. The combined SX–EW process was barely commercialized by 1971, but already accounts for 27 per cent of US copper mine output, and about 12 per cent of world output. (See, for instance, the chapter on copper in US Bureau of Mines *Minerals Yearbook*, 1989). Capacity is expected to double by the year 2000.

Similarly, sulfur dioxide, carbon monoxide and carbon dioxide are needed for certain chemical synthesis processes, but these chemicals cannot be economically transported more than a few kilometers at most. Hydrogen, produced in petroleum refineries, can be compressed and shipped but it is much better to use it locally. A hydrogen pipeline network has been built for this purpose in the Ruhr Valley of Germany. So-called 'blast-furnace gas' can be burned as fuel, but it is not economical to transport very far. There are numerous examples in the chemicals industry, especially. Closing the materials cycle can take the form of creating internal markets (uses) for low-value by-products by upgrading them to standard marketable commodities. (This often depends, incidentally, upon returns to scale, but only in a particular context.)

To summarize, there are significant potential returns to internalization of the materials cycle. For instance, the so-called 'integrated' steel mill (including its own ore sintering and smelting stages) is an example. It would not pay to produce pig iron in one location and ship it to another location for conversion to steel, for two reasons: (1) there would be no way to use the heating value of the blast furnace gas and (2) the molten pig iron would cool off en route and it would have to be melted again. Thus energy conservation considerations, in this case, dictate integration. The same logic holds for petroleum refineries and petrochemical complexes. In each case there are a number of low-value intermediate products that can be utilized beneficially if, and only if, the use is local. However, these examples of integration obviously require a fairly large scale for viability. This is in direct contrast to the strategy of 'end-of-pipe' waste treatment and disposal, which is normally practiced in smaller operations.

INTEGRATED INDUSTRIAL ECOSYSTEMS

At this point it is probably useful to introduce the notion of industrial ecology (IE) more formally. Industrial ecology is a neologism intended to call attention to a biological analogy: the fact that an ecosystem tends to recycle most essential nutrients, using only energy from the sun to drive the system.[1] The analogy with ecosystems is obvious and appealing (Ayres 1989a). In a 'perfect' ecosystem the only input is energy from the sun. All other materials are recycled biologically, in the sense that each species' waste products are the 'food' of another species. The carbon–oxygen cycle exemplifies this idea: plants consume carbon dioxide and produce oxygen as a waste. Animals, in turn, require oxygen for respiration, but produce carbon dioxide as a metabolic waste. In reality, the biosphere does *not* recycle all of the important nutrient elements – notably phosphorus and calcium – without help from geological processes; but this is probably a quibble. An ecosystem involves a 'food chain' with a number of interacting niches, including primary photosynthesizers (plants), herbivores, carnivores preying on the herbivores, saprophytes, parasites and decay organisms.

The idea of 'industrial ecology' has taken root in the past few years, especially since the well-known article by Frosch and Gallopoulos in a special issue of *Scientific American* (Frosch and Gallopoulos 1989). The industrial analog of an ecosystem is an industrial park (or some larger region) which captures and recycles all physical materials internally, consuming only energy from outside the system, and producing only non-material services for sale to consumers. This vision is highly idealized, of course. The notion of deliberately creating 'industrial ecosystems' of a somewhat less ambitious sort has become increasingly attractive in recent years. Author Paul Hawken has commented: 'Imagine what a team of designers could come up with if they were to start from scratch, locating and specifying industries and factories that had potentially synergistic and symbiotic relationships' (Hawken 1993 p. 63).

An industrial ecosystem, then, could be a number of firms grouped around a primary raw material processor, a refiner or convertor, and a fabricator, various suppliers, waste processors, secondary materials processors, and so forth. Or it could be a number of firms grouped around a fuel processor, or even a waste recycler. The main requirement is that there be a major 'export product' for the system as a whole, and that most of the wastes and by-products be utilized locally.

But, as this book has pointed out, there are in fact a large number of plausible possibilities for 'internalizing' material flows in various ways. This possibility is not restricted to process wastes from industry. It also applies to final consumption wastes, for example of packaging materials. At the industrial level, this implies that some firms must use the wastes from other firms as raw material feedstocks. Other firms must use the wastes from final consumers in a similar manner. The complex web of exchange relationships among such a set of firms can be called an 'industrial ecosystem'. This concept was given a considerable boost in recent years by Robert Frosch, especially in the article in *Scientific American* cited above (Frosch and Gallopoulos 1989). With others, he has also encouraged the US National Academy of Engineering to sponsor a series of summer studies (leading to books) promoting the concept and exploring various aspects.

At first glance, systems integration looks rather like old-style 'vertical integration', except that there is no need for all of these enterprises to have common ownership. In fact, the flexibility and innovativeness needed for long-term success is more likely to be promoted by dispersed ownership. Yet a considerable degree of inter-firm cooperation is needed as well. We will return to this point later.

The most influential – and possibly the only – prototype for such a system was, and still is, the Danish town of Kalundborg. In this town waste heat from a power plant and a petroleum refinery has been used to heat greenhouses and other wastes from several large industries have been successfully converted into useful products such as fertilizer for farmers, building materials, and so on. The Kalundborg example is discussed in full in Chapter 33.

OTHER POSSIBLE INDUSTRIAL ECOSYSTEMS

Several proposals superficially comparable to the Kalundborg example have been made. One of the oldest is the 'nu-plex' concept, promoted vigorously by nuclear power advocates at Oak Ridge National Laboratory (USA) in the 1970s. It was, however, basically an idea for an industrial park for large-scale electric power consumers.

A more interesting scheme, from our perspective, is a proposal *aluminum-kombinat* for utilizing low-grade (high-ash content) anthracite coal to recover aluminum and cement (Yun *et al.* 1980). The project was conceived at the Korean Institute of Science and Technology (KIST) as a possible answer to two problems. First, the city of Seoul needed to dispose of several million metric tons of coal ash each year. At the same time, South Korea was totally dependent on imported aluminum, and there was a strong desire to become self-sufficient. After several years of investigation, the *kombinat* scheme evolved. As of 1980, a 60 metric ton per day pilot plant was in operation and process economics appeared to be favorable.

In brief, the energy from coal combustion would be used to generate the electric power for aluminum smelting. The inputs to the *kombinat* would be low sulfur anthracite coal (1.9 million metric tons per year – MMT/yr), limestone (3.9MMT/yr) and clay (0.48MMT/yr). Outputs would be 100 thousand metric tons (100 kMT) of aluminum and 3.5MMT of Portland cement. The heart of the scheme is an alumina plant, consisting of two units: a sintering plant (coal + limestone + soda ash) yielding high-temperature exhaust gases (900 °C) for the steam turbine and 2MMT/yr clinker for the leaching unit. The latter grinds the clinker and leaches the alumina with hot sodium carbonate solution. The soda combines with alumina, yielding sodium aluminate in solution, while the lime combines with silica precipitating as dicalcium silicate. The latter is sent to the cement plant. The sodium aluminate is then treated in a conventional sequence, first by adding lime to precipitate the dissolved silica and then carbonation of the solution (with CO_2 from the waste heat boiler) to reconstitute the soda ash and precipitate aluminum hydroxide. When aluminum hydroxide is dehydrated (that is, calcined) it becomes alumina. About 40kMT/yr of soda ash would be lost in the soda cycle, and would have to be made up. According to calculations and test results, aluminum recovery from the ash would be about 71 per cent, while the thermal efficiency of the electric power-generating unit would only be about 15 per cent owing to the considerable need for process steam by the leaching plant, mostly for calcination. The basic scheme is outlined in Figure 5.1.

A scheme similar to the *kombinat* was analyzed independently in the late 1970s by TRW

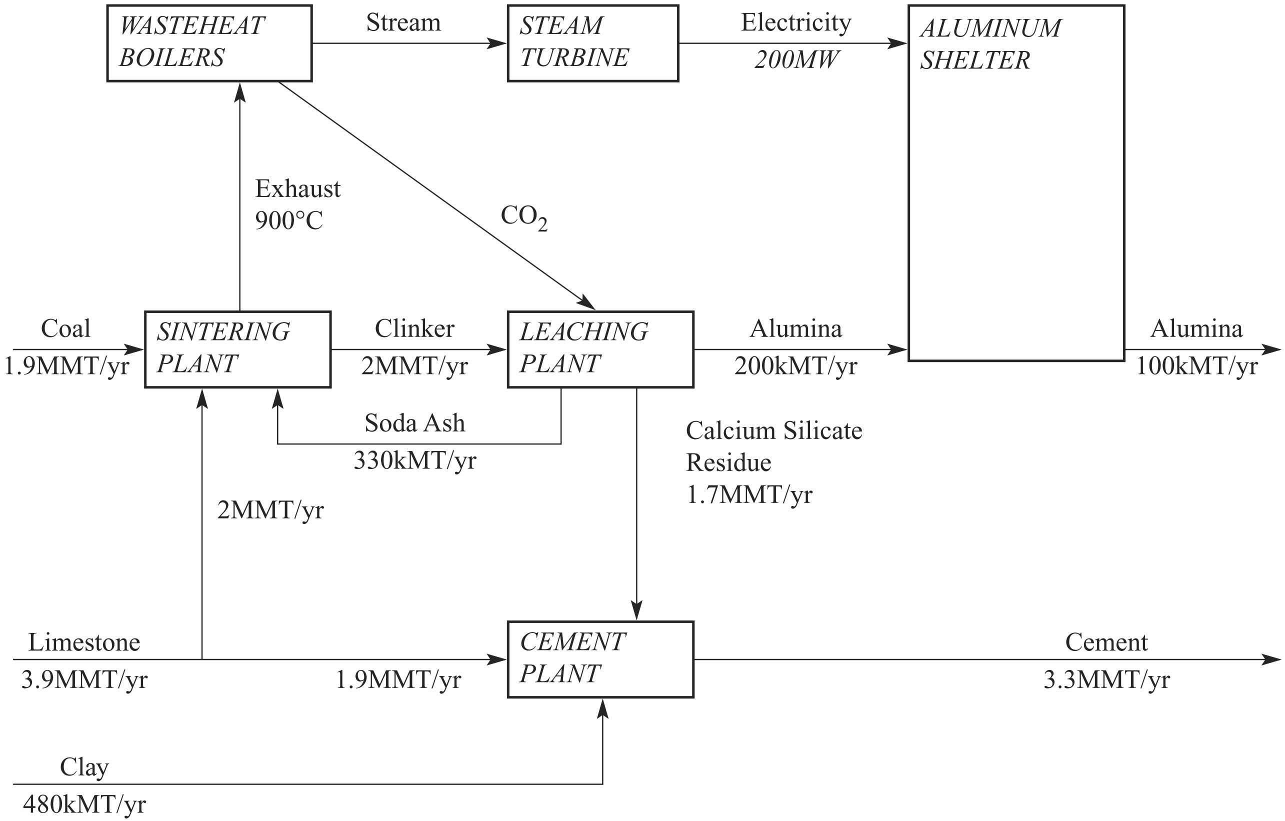

Figure 5.1 Conceptual diagram of an aluminum kombinat

Inc. for the US Environmental Protection Agency (Motley and Cosgrove 1978). The idea was motivated by the fact that flue gas desulfurization (FGD) technology was just being introduced by coal-burning electric power plants. The technology then being adopted was lime/limestone scrubbing, which captures sulfur dioxide quite effectively but generates large quantities of calcium sulfite/sulfate wastes. The TRW study evaluated a possible use for these wastes.

The scheme was based on a conceptual coal-burning power plant generating 1000MW, which generates 1MMT/yr of lime/limestone scrubber wastes. The core of the scheme would be a sinter plant in which the sulfate sludges react with carbon monoxide produced by burning coal (273kMT), clay (300kMT/yr) and soda ash (12kMT/yr), to yield soluble sodium aluminate, dicalcium silicate and hydrogen sulfide. These, in turn, are processed by standard means (indicated briefly in the description of the *kombinat* above), to yield calcined alumina (70kMT), elemental sulfur (156kMT) and dicalcium silicate (625kMT). The latter, in turn, is the major ingredient to produce 850kMT of Portland cement. At typical market prices, this scheme appeared to be viable, or nearly so. It would certainly be viable given a realistic credit for FGD waste disposal.

Another interesting proposal for an industrial ecosystem comes from Poland (Zebrowski and Rejewski 1987). It is actually a set of interrelated proposals utilizing two basic technologies that have been under development in Poland. The first is coal pyrolysis in the gas stream (PYGAS), a patented technology,[2] that has already been adopted at several Polish industrial sites. It is particularly suited to upgrading existing power plants at minimal capital cost. The basic idea is to feed powdered coal into a hot gas stream (about 800°C) where it pyrolyzes very rapidly (in the order of one second), and pyritic sulfur also decomposes at this temperature. The gas stream passes through a cyclone, where desulfurized carbon char dust is collected and removed. It is usable as a direct substitute for powdered coal in the boilers. Some of the gas is recycled. The pyrolysis gas can be desulfurized and burned or used as feedstock for chemical processing. The second building block is a technology derived from PYGAS for pyrolysis of recycled gas streams, PYREG, specialized to the case of lignite. It has been developed to the large-scale laboratory test stage at the Industrial Chemistry Research Institute (ICRI) in Warsaw.

The idea is not qualitatively different from numerous other proposals for coal gasification, but the authors have given careful consideration to the use of these technologies to integrate existing disconnected systems, especially with respect to sulfur recovery and fertilizer production. This concept is called the Energo-Chemical PYREG site, or simply ENECHEM. The base case for comparison would be a surface lignite mine (18MMT/yr), with 0.5 per cent sulfur content. This would feed a power station generating 2160MW of electricity. Lignite in Poland (and central Europe generally) contains 2 per cent –10 per cent xylites (5 per cent average). Xylites are potentially useful organic compounds related to xylene ($C_6H_4(CH_3)_2$), which are not recovered when lignite is simply burned.

In the base case, annual wastage of xylites would be 900kMT. By contrast, PYREG technology permits the direct recovery of xylites in the form of high-grade solid fuel (semicoke, 200kMT/yr), fatty acids and ketenes (65kMT/yr) and gaseous aromatics (benzene, toluene, xylene or BTX), which are normally derived from petroleum refineries. The proposed ENECHEM site would include a power station, but instead of burning lignite directly to generate 2160MW as in the base case, it would gasify the lignite, via PYREG, as shown in Figure 5.2, yielding semicoke powder plus volatile hydrocarbons, tar and phenolic water. The semicoke powder would then be burned in the power station (generating 1440MW, and emitting about 97kMT/yr SO_2). Sulfur recovery in PYREG technology is only 40 per cent–60 per cent, but since Polish lignite has a very low sulfur content (0.5 per cent) this is not considered to be a major disadvantage.

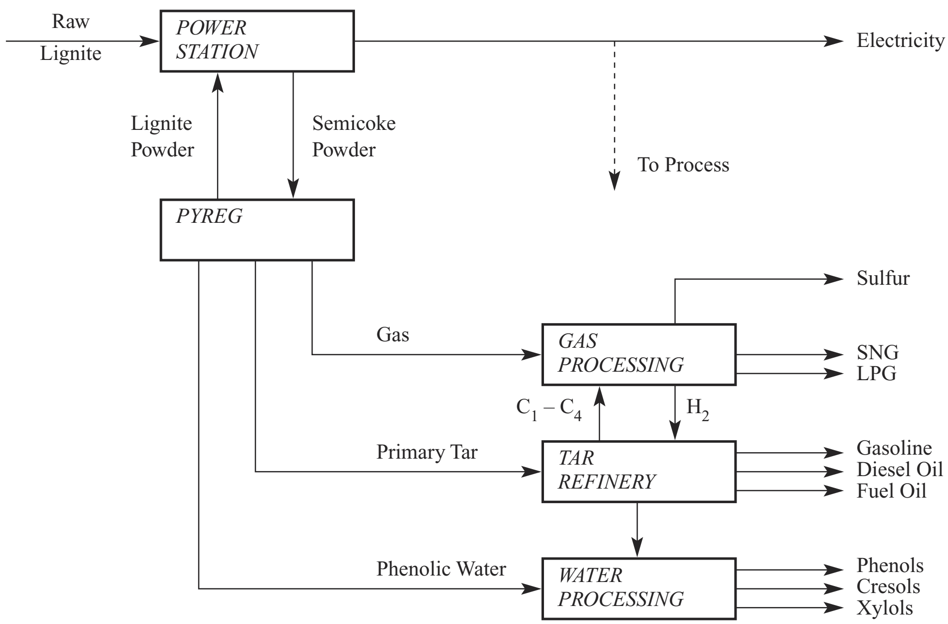

Figure 5.2 Lignite-burning power plant modified via PYREG

The volatile hydrocarbon fraction of the PYREG output would be desulfurized – by conventional Claus technology – yielding about 67kMT/yr of elemental sulfur (S). The condensibles would be separated as liquid propane gas (LPG) for domestic use (150kMT/yr). The non-condensibles, consisting of methane and ethane or synthetic natural gas (SNG), would be available as a feedstock to any natural gas user, such as an ammonia synthesis plant (318kMT/yr). The tar from the PYREG unit could be refined much as petroleum is, yielding liquid fuels and some light fractions (C_2–C_4) that would go to the gas processing unit. The yield of gasoline and diesel oil would be 430kMT/yr and 58kMT/yr, respectively, plus 75kMT/yr of heavy fuel oil. (Obviously, the tars could be a supplementary feed to a co-located conventional petroleum refinery, but the incremental outputs would be much the same.) The phenolic water would be processed to recover phenols (13kMT/yr), cresols (27kMT/yr) and xylols (26kMT/yr).

Obviously, the details of ENECHEM could be varied considerably, but the scheme as outlined in the previous paragraph would reduce sulfur dioxide emissions by roughly half (from 180kMT to 97kMT). It would produce less electric power but, in exchange for a reduction of 720MW, it would yield 318kMT (400 million cubic meters) of SNG, 150kMT LPG, 430kMT gasoline, 480kMT diesel fuel, 75kMT heavy fuel oil (less than 1 per cent S), 66kMT of phenols, cresols and xylols, and 67kMT of sulfur (99.5 per cent). This is shown in Figure 5.3.

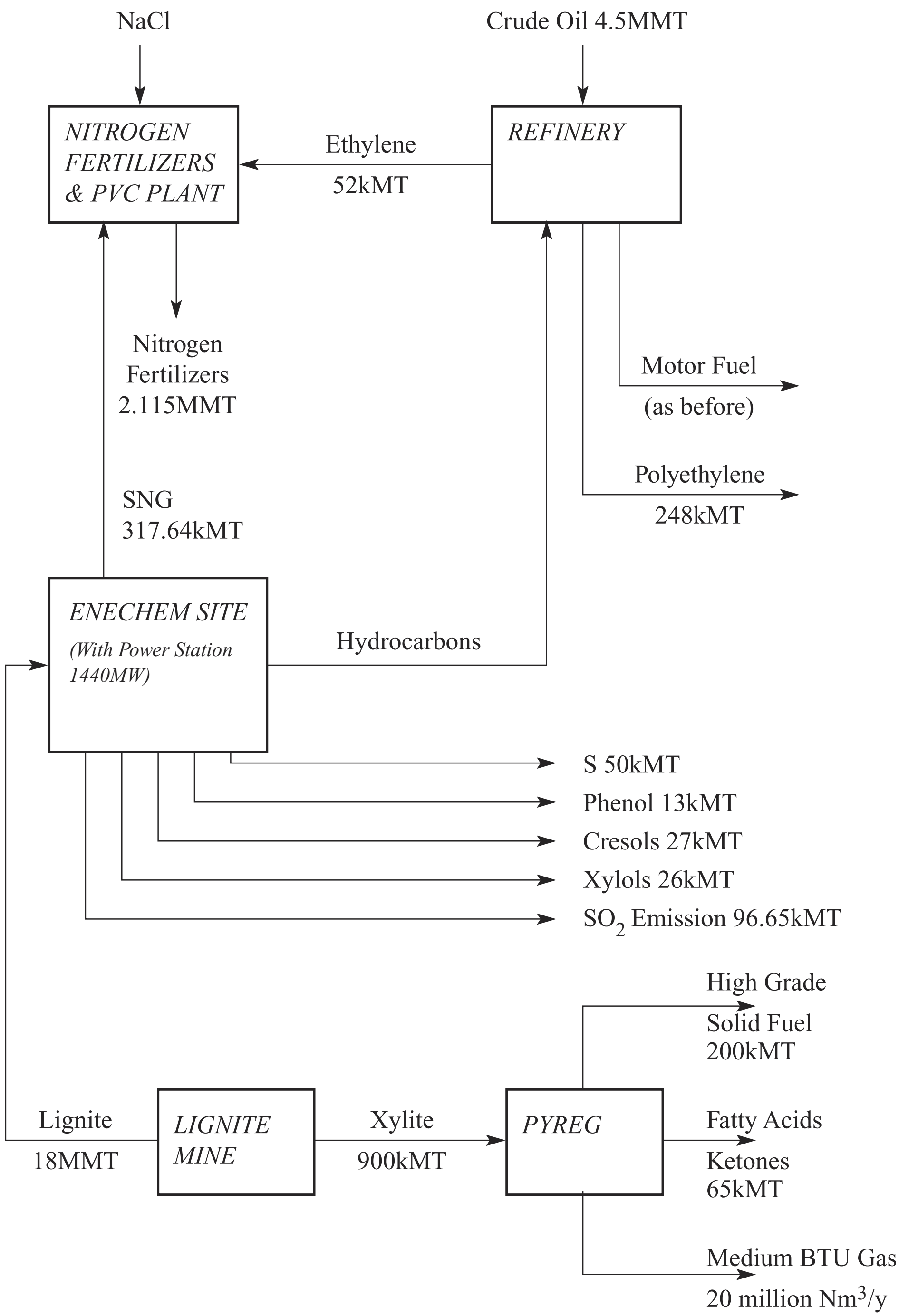

Figure 5.3 Systems integrated with ENECHEM with additional plant for xylite processing

A recent proposal by Cornell University to the US Environmental Protection Agency differs sharply from the schemes outlined above. Instead of focusing on utilizing an existing natural resource more efficiently, it would attempt to assemble the elements of an industrial ecosystem around a municipal waste treatment facility. In other words, it is essentially a scheme to 'mine' wastes per se. The proposal points out that in the 1970s a number of facilities were built with the idea of reducing landfill volumes by recovering the combustible fraction, along with ferrous metals, and converting it to a refuse-derived fuel (RDF), to be sold to a local utility to help defray the costs of operating the facility. Many of these facilities operated only briefly or not at all.

The Cornell proposal would extend the earlier waste treatment concept in two ways. First, it would include not only municipal wastes but also a variety of other industrial wastes for a whole county. Second, it would employ advanced technologies to produce a number of salable by-products, one of which would be fuel gas. (Nevertheless, its success would still depend on one or more utilities that would undertake to accept the gaseous fuel generated.)

Also, in contrast with other schemes outlined in this chapter, it would involve no detailed prior planning of the site or the technology to be used, beyond the creation of an organizational structure to seek out potential participants. This approach is almost mandatory, at least in the USA, where central planning is virtually anathema today. Nevertheless, the proposal (if supported) would offer some useful insights as to how a cooperative entity might be created from essentially competitive, independent production units – or, indeed, whether this is possible.

A final example might be COALPLEX, first proposed by the author some years ago (Ayres 1982) and revised more recently (Ayres 1993c, 1994b). It too would be coal-based. Like the Polish scheme it would start with gasification of the coal, recovering sulfur for sale and using the coal ash as a source of alumina (and/or aluminum) and ferrosilicon (Figure 5.4). The most attractive version – albeit somewhat theoretical – would utilize the direct (hydrochloric) acid leaching process for aluminum chloride recovery and the ALCOA process for electrolysis of the chloride. The gasified coal would be (partly) burned on site to produce electric power for the aluminum smelter and electric furnaces. A variant would also produce carbon anodes for the aluminum smelter from coke made from gasified coal (instead of petroleum coke). There are, in fact, a number of possible variants, none of which have been adequately analyzed to date.

CONCLUDING COMMENTS

The key feature of most industrial ecosystems that have been proposed is what might be termed 'economies of integration'. To be sure, large scale is also required, in most cases. But beyond that, both vertical and horizontal integration are required. All industrial ecosystems essentially depend on converting (former) waste streams into useful products. This means that some producers must be induced to accept unfamiliar inputs (that is, converted wastes) rather than traditional raw materials. In some cases they will have to invest large sums of money to create new processing facilities, based on unproven – or semi-proven – concepts.

An industrial ecosystem must look like a single economic entity (firm) from the outside.

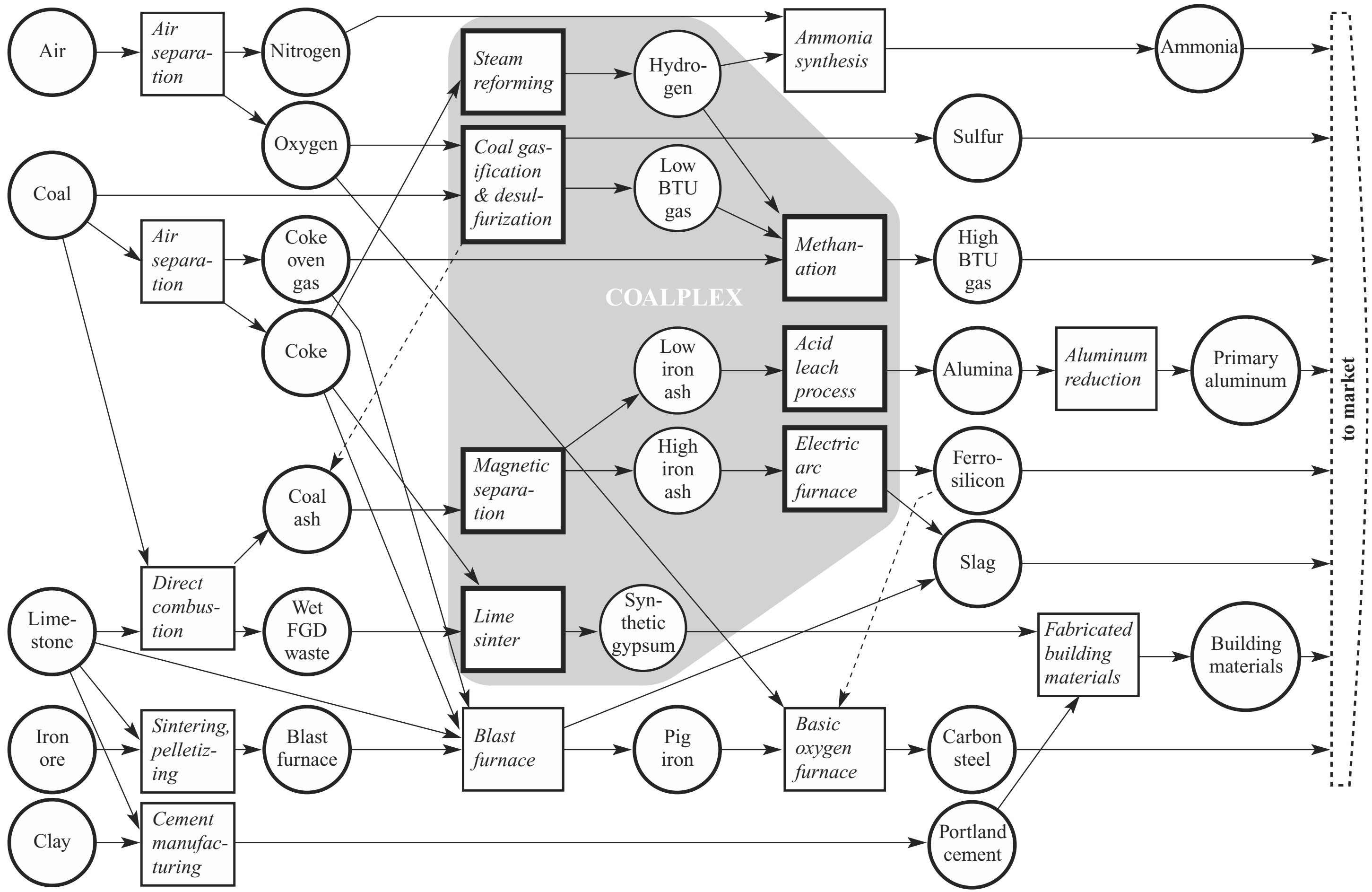

Figure 5.4 Hypothetical process–product flows for COALPLEX

It will have consolidated inputs and outputs (products). It will compete with other such entities (firms) in both raw material and product markets. It will also compete with other firms for capital. From the inside, however, central ownership with hierarchical management is almost certainly not the optimum solution. Too much depends on very sensitive and continuous adjustments between the different components of the system. This is much more compatible with markets – provided all parties have relatively complete information – than it is with centralized top-down management (as critics of Taylorism have been saying for a long time). Yet the modern version of a conglomerate, consisting of autonomous units linked to a corporate parent by a purely financial set of controls, with each component competing for funds on the basis of profits, cannot work either.

Evidently, an industrial ecosystem based on a major primary 'exporter' with a galaxy of associated waste convertors is quite unlike a traditional impersonal 'market', where goods of known and constant quality are bought and sold through intermediaries. In a traditional market there are many competing suppliers and many competing consumers. Because only high-value goods are physically shipped over distances, many suppliers need not be local. Thus traditional industrial systems can be quite decentralized. Indeed, the pattern established by many multinational manufacturing firms is to build various elements of a product line in several different countries, thus achieving both the benefits of scale, on the one hand, and minimizing the risk of being 'held up' by local governments or unions, on the other. Ford's 'world car' concept is a good example, although IBM was (is) perhaps the most successful practitioner of the policy of decentralized international supply.

In an industrial ecosystem, for obvious reasons, low-value materials from a first-tier exporter must be utilized locally. Assuming there is one major waste, that can be converted into a useful raw material, there must also be a local user for that raw material. This is likely to be a fairly large firm, also, to achieve the necessary scale of operation. Other satellite firms in the complex will have only one supplier for a given input, and one consumer for a given output.

The necessity for close long-term cooperation and planning between waste producers and consumers is obvious. Neither can change either processes or production levels without strongly affecting the other. Less obvious, but not less important, is the corresponding need for relatively complete disclosure of all relevant technological information on both sides. Being accustomed to a culture of secrecy, this condition will be very difficult for most firms to achieve. In general, it appears that an enforcement mechanism or some economic inducement will be needed to ensure cooperation.

There are three existing models for a cooperative system. One is the 'common ownership' model of vertical integration, in which a single corporate entity owns all, or most, of its suppliers and manages the whole collection centrally. AT&T, GE, GM, IBM, Standard Oil (NJ) and US Steel were created by multiple mergers to dominate an industry, and mostly followed this pattern. ALCOA, Ethyl Corp and Xerox achieved monopoly status originally through tight control of patent rights, but both Ethyl and Xerox lost dominance when the patents expired. In ALCOA's case, as with AT&T and Standard Oil, the monopoly was broken by anti-trust action by the US government. But, as the example of GM illustrates, vertical integration is no longer necessarily an advantage. In fact, GM's principal corporate disadvantage, *vis-à-vis* its competitors, is that it obtains more of its components from wholly owned subsidiaries that are forced to pay high wages negotiated with the United Auto Workers Union, than do other auto firms; the Japanese companies are the least vertically integrated of all. But perhaps the major problem with vertical integration is that it is too cumbersome. IBM has lost ground to a number of smaller, nimbler, competitors mainly because decision making is too centralized and slow.

The second model for inter-firm cooperation is most familiar in Japan, where it is known as the *keiretsu.* The *keiretsu* is a family of firms, normally controlled indirectly by a large bank, with links to a common trading company, and several major first-tier manufacturers spread over a range of industries. There is also, typically, a collection of smaller satellite suppliers for each first-tier company. Most of the larger firms in the group have interlocking stockholdings, and the smaller firms tend to be controlled by the larger ones who are their customers. Thus long-term relationships are essentially guaranteed by financial means.

The diversified-portfolio conglomerate, as exemplified in the USA (for example, ITT, LTV, Textron, Berkshire-Hathaway, Seagrams) offers a superficially different scheme. In principle, it could also be a mechanism for obtaining inter-firm cooperation by means of financial controls. The diversified portfolio version rarely attempts any such thing, however. On the contrary, these entities are normally created by financiers in order to exploit 'synergies' (that often turn out to be illusory) and little or no actual cooperation at the technical level takes place. The AT&T purchase of NCR, which had a technological justification, turned out no better. But none of these examples seems ideal to fit the needs of an industrial ecosystem.

The third model for inter-firm cooperation is also top-down, usually being organized by marketing organizations. Families of (largely unrelated) suppliers with long-term contracts have been created by a number of large retail marketing firms, such as Sears-Roebuck, Wal-Mart, K-Mart and McDonald's. Normally, however, each supplier manufactures a different product, so there is actually little need for technical cooperation among them. But, in at least one case, the supplier family evolved from a large-scale manufacturer. The traditional Italian textile industry, originally a group of large multi-product companies in a declining industry, has radically reconstructed itself by a policy of deliberate functional spinoffs, leaving many small subcontractors to a dominant marketer (Benetton). This self-induced change has been extraordinarily successful. However, it is not particularly relevant to the problem of creating industrial ecosystems.

An industrial ecosystem could theoretically be created by an actual merger of many existing firms at a single location, to promote the necessary inter-firm cooperation. But this is unlikely to make sense, except in very rare instances. More likely, the necessary cooperation could be induced by a local government or some other public body, as seems to have been the case in Kalundborg. On the other hand, no public entity in any of the major industrial countries (to our knowledge) has yet moved beyond the concept stage.

One of the most ambitious such concepts is the 'Eco-Park', proposed by a group at Dalhousie University, Nova Scotia (Coté *et al.* 1994). The basic idea is to convert an existing, traditional industrial park – the Burnside Industrial Park in Dartmouth, Nova Scotia – into an industrial ecosystem. The Burnside Industrial Park already has a base of 1200 businesses, but the land was assembled and provided by the city, with infrastructural funds provided by the central government of Canada through the Atlantic Development Board, a regional development agency. The city still retains approximately 3000 acres. Thus public agencies retain very significant influence over the future evolution of the area.

However, it is unclear just how this potential influence can be brought to bear to create the necessary incentives *and mechanisms* to create a successful Eco-Park.

Each component of an industrial ecosystem has its unique role to play. Financial allocations among components cannot be made on the basis of 'profits', for the simple but compelling reason that some components will do business only with others, and there is no unique or objective way to set meaningful prices for all internal transactions. (Companies do it in a variety of ways – often to minimize taxes – but all have their disadvantages and critics.) Only by comparing different internal technology choices and transactional arrangements in terms of their impact on the external competitiveness of the system *as a whole* can objective choices be made. Moreover, such comparisons are necessarily dynamic, rather than static, which makes the decision process very complex indeed.

To summarize, industrial ecosystems are very appealing in concept. Properly organized and structured, they exemplify a built-in incentive to minimize wastes and losses of intermediates. But much research is needed to clarify the optimum organizational and financial structure of such an entity.

NOTES

1. Thus a natural ecosystem is a self-organizing system consisting of interacting individuals and species, each programmed to maximize its own utility (survival and reproduction), each receiving and providing services to others, each therefore dependent on the system as a whole. The ecosystem normally maintains itself in a balanced condition, or evolves slowly along a developmental path. But such dissipative systems remain far from (thermodynamic) equilibrium.
2. Patent no. 87904. License available from PROSYNCHEM Design Office, Gliwice, Poland.

6. Industrial ecology: governance, laws and regulations

Braden R. Allenby

Industrial ecology deals for the most part with environmental science, and technology and technological systems, but these do not exist in a vacuum. Thus the industrial ecologist should be familiar not just with the techniques and principles of the field, but also with the cultural and legal context within which they are embedded. These dimensions are usually interrelated with economic and other policy issues (see Chapter 5). Taken together, these dimensions are integrated in general policies, practices, laws and regulation that vary widely between jurisdictions. Rather than focus on specifics that may be relevant only in particular jurisdictions, therefore, this chapter will present a general introduction to the governance and legal contexts within which industrial ecology issues are likely to arise and be resolved.

In many fields, a discussion of law and regulation is straightforward, if detailed. Industrial ecology, however, offers a more daunting analytical challenge, for two principal reasons. First, it represents the evolution of environmental policy from overhead to strategic for both society and firms. As overhead, environment was essentially an afterthought, to be taken care of once the core activity, whether it was producing widgets in the firm, or carrying out national security policy as a nation state, was already done. For example, putting scrubbers on a manufacturing facility is an overhead approach; indeed, such environmental expenditures appear in corporate accounting systems in the overhead accounts (Todd 1994). Designing a personal computer to be cost-efficient in a jurisdiction that requires product 'take back', however, is a strategic function (Graedel and Allenby 1995). In the USA, for example, routinely assigning all 'environmental' issues to the US Environmental Protection Agency, regardless of what underlying governmental function was involved, is an indication that such issues were regarded as overhead. But this is shifting, as the current dialogues (indeed conflicts) involving the environmental community, and other policy structures such as trade or national security, illustrate (Allenby 1999a, pp. 6–9; Allenby 2000a). This difficult adjustment period is to be expected as policy communities that have hitherto been separate – as, for example, the environmental community and the national security community – attempt to work with each other to achieve integrated approaches.

Additionally, it is now apparent that the environmental perturbations of major concern, such as global climate change, loss of biodiversity, degradation of oceanic and water resources, are transboundary in nature. They do not reflect, or respect, human jurisdictional demarcations. This makes industrial ecology, which deals with the relationships among such systems and related human systems, particularly sensitive to jurisdictional effects and prevalent global governance structures. Thus the fact that the current international political structure that has dominated the world for hundreds of years, predicated on the absolute sovereignty of the nation state, is currently in a state of flux is an important dynamic for industrial ecology. Indeed, the outlines of a new, more complex, international governance system are emerging, and it is important that the industrial ecologist be comfortable with this development.

Finally, the theoretical foundations upon which robust industrial ecology policy structures could be based do not yet exist. The theory of technological evolution is underdeveloped (Grübler 1998), management systems that are adaptive enough to manage complex resource systems effectively do not exist (Gunderson *et al.* 1995; Berkes and Folke 1998) and the challenge of earth systems engineering and management (see Chapter 46; also Allenby 1999b) is one of which people are just becoming aware. This indication of our ignorance should encourage a certain humility and diffidence in approaching this subject.

GLOBAL GOVERNANCE SYSTEMS

To begin with, there is a difference between 'governance' and 'government'. 'Governance' is the process by which society at different levels is managed and administered. It generally consists of both implicit and explicit practices, relationships and structures, and is frequently difficult to define with precision. 'Governments' are the formal institutions which administer laws and regulations, and maintain civil order, over states, districts, localities, cities, towns and other political entities. Governments play important roles in governance, but governance as a function is broader than just the activity of governments. Since the treaties of Westphalia in 1648, the traditional global governance structure has been based for hundreds of years on the institution of the nation state: examples are the USA, China, France and Brazil (Cooper 1996; Mathews 1997; *The Economist* 1997). Thus, under traditional international law, the only entities that are considered competent to make treaties, negotiate agreements and represent citizens in international fora are countries. For example, the negotiations about global climate change mitigation measures are conducted entirely by nation states, although firms and environmental non-governmental organizations (NGOs) are able to participate and lobby behind the scenes. But this governance system has become much more complex over the past decade. As Figure 6.1 illustrates, where the nation state used to be dominant, it now is just one of many institutions involved in international governance . Private firms, NGOs and communities of different kinds now increasingly share responsibility for international policy development and implementation (Mathews 1997; *The Economist* 1998, especially p. 16: '[state] sovereignty is no longer absolute, but conditional'; *The Economist* 2000). Formal practice has yet to catch up with this new reality, nor are any of these entities clear about their roles in the still evolving governance structures, but the practicing industrial ecologist cannot afford to ignore it.

There are several reasons for this evolution of international and regional governance structures. First, transnational corporations have grown much larger, to the point where their financial power equals that of many small countries. This financial power has been augmented because private firms, by and large, are the repository of technological sophistication in society, so to the extent that solutions to environmental and human rights

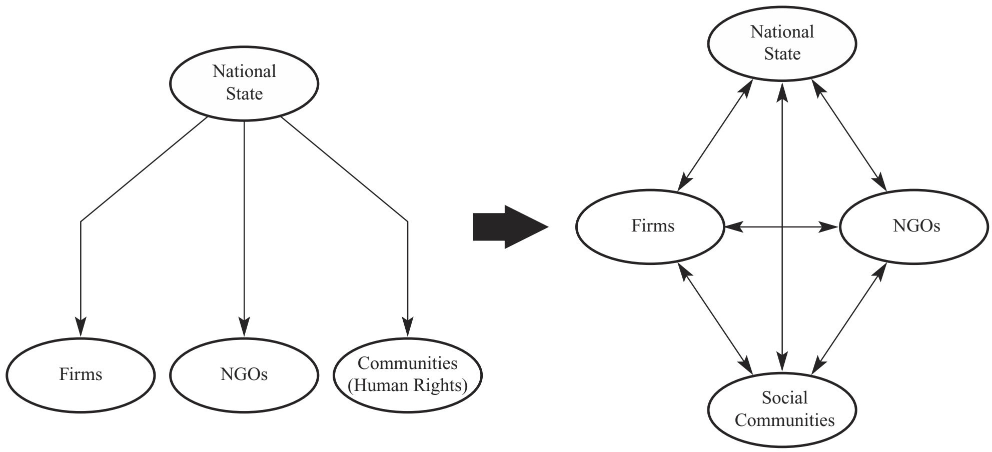

Figure 6.1 Evolution in international governance systems

issues involve technology, they involve private firms as primary actors. Thus, for example, governments can ban chlorofluorocarbons (CFCs) to protect stratospheric ozone, but it is generally private firms that develop and deploy substitute technologies and, ultimately, ensure that such a policy is technologically practicable. It is thus increasingly clear that the mitigation of environmental perturbations requires the partnership of private firms (Netherlands Ministry of Housing, Spatial Planning and the Environment 1994a; Grübler 1998).

The increasing importance of firms is balanced by that of non-governmental organizations, or NGOs, which have also grown in power and civic authority. In fact, a number of governments, especially in Europe, use NGOs to perform many functions that they would have performed themselves in the past, such as distributing food aid in African countries stricken by drought. Many of the significant environmental and social conflicts of the late 20th century, such as the radical response to genetically modified organisms in Europe, the human rights confrontations over working conditions in Asian facilities, and the sometimes violent attacks on trade and international financial institutions, have not involved nation states, but NGOs. Polls routinely show that NGOs have more credibility on environmental issues than scientists, private firms or even government environmental regulators. NGOs are different from other actors in the global governance system in two important ways. First, many such groups, reflecting their informal and populist origins, tend to be issue-specific. Second, unlike the case with nation states or firms, there are no institutional safeguards regarding the establishment of NGOs. Thus virtually anyone can set up an NGO to represent almost any position within the applicable constraints of the laws of the jurisdiction. While this is very democratic, it also means that there are few governance mechanisms for NGOs, or controls, should some of them choose to act irresponsibly (*The Economist* 2000).

The importance of communities has also increased. These communities are generally of two types, those that are defined geographically, and those that are defined by interests. There are a number of places in the world, particularly in Africa, where the nation state structure has not taken hold (Cooper 1996), and in those areas community, rather than nation state, representatives may more validly reflect the interests of the citizens. Elsewhere, communities that are especially affected by certain phenomena, such as siting of toxic waste dumps, may want to participate in governance dialogues because they believe that their interests are not being adequately protected by other participants. In addition, the growth of the internet and communications infrastructure has made it much easier for communities of interest to consolidate around issues, and represent themselves forcefully to other participants in the governance process.

One important facet of this change in global governance structure is that it is not leading to a world that is more homogeneous. 'Globalization', whether cultural, economic or institutional, is indeed a powerful current trend, but it is not necessarily a simplifying trend. It is not a case of 'either globalization or localization'; instead, global and local economic and cultural interests are developing simultaneously. What is really happening is that the world is becoming far more complex, both more local and more global; simpler in some ways, much more complicated in others (Mathews 1997; Sassens 1996; Watson 2000; Allenby 2000/2001). This increasing complexity is also reflected in the rise of post-modernism, with its emphasis on visual and intellectual pastiche and encouragement of multicultural discourses (Berman 1982; Harvey 1996; Anderson 1998). It is in this system of increasing economic, cultural and institutional complexity, marked by still evolving global governance systems which are themselves increasingly complex, that the student of industrial ecology must function.

GOVERNMENT STRUCTURE AND CULTURE

Law and regulation, and the means and practices by which they are implemented, are expressions of the jurisdiction and, more fundamentally, the culture. Thus it is important for the student of industrial ecology to recognize that governments, especially at the nation state level, differ along a number of dimensions that can significantly affect their ability to respond to environmental challenges. Among the most important of these (Graedel and Allenby 1995; Allenby 1999a) are the following:

1. The form of government: in general, democracies such as those in Western Europe and the USA tend to be more responsive than more totalitarian governments such as those that used to exist in Eastern Europe.
2. Wealth: wealthier countries have more resources to respond to environmental challenges than do poorer countries, and the former may be able to place more relative value on environmental benefits. In addition, level of national wealth determines the type of environmental issue which is likely to be of concern: in a developing country, the most pressing environmental issues may be sanitation and safe water in urban areas, whereas a developed country may be more able to turn its resources to enhanced water quality in major watersheds or global climate change issues (World Bank 1992b).
3. Size of country: even very progressive small countries such as Denmark or the Netherlands cannot overlook the fact that much of their industrial production is exported, and thus subject to standards and requirements beyond their direct reach.
4. Focus: countries emphasize different aspects of environmental protection. For

example, the USA tends to be a leader in remediation but lags behind Japan in energy efficiency and behind Germany and the Netherlands in developing consumer product 'take back' approaches (see Chapter 40). National cultures and technological trajectories are important determinants of such patterns (Grübler 1998).

5. Culture: there is a distinct contrast between, for example, Japan with its parsimonious approach to resources and energy born of its island status, and the former Soviet Union. The latter possessed a greater natural resource base and a focus on industrialization at any cost that created an accompanying cavalier attitude towards conservation.
6. Attitude towards formal law: some countries, such as the USA, have a law-based culture where equality before the law, and clarity of application, dominate interactions, especially economic ones. Other countries have more informal systems, where written law is only one of a number of considerations, including kinship, which affect commercial and institutional relationships. The relationships between, and relative importance of, formal and informal legal structures is an important characteristic of any culture.

LEGAL CONSIDERATIONS

Given that the dimensions discussed above will create different policy spaces for different jurisdictions and cultures, it is nonetheless the case that there are certain fundamental legal principles and issues the student of industrial ecology should be aware of. Although perhaps in different guise, these come up with regularity in any effort to generate policies in the area of industrial ecology. Among the most important are the following:

1. The issue of intragenerational and intergenerational equity is important, especially as an egalitarian distribution of wealth and resources within and among generations is a key element of the concept of sustainable development (WCED 1987; Weiss 1989). The distribution of wealth and power both among nation states and between the elites and the marginalized populations within individual nation states is of course one of the thematic foundations of political science. It is also a highly ideological and contentious arena, where the interplay between law and culture is particularly charged. To the extent that any policy, such as sustainable development, implies a substantial shift in resources between rich and poor nations, as well as within nations, it is always controversial.
2. The issue of whether, and how, future generations can or should be given rights in existing legal proceedings is a difficult one. Rights only arise when there are identifiable interests, and it is almost by definition impossible to identify either future individuals or their interests with sufficient specificity to involve them in adjudication of such interests. Who, for example, knows what resources will be critical to future technologies? Who can say for sure what the preferences of future generations will be? Assuming that some degree of intragenerational inequality still exists, whose interests will be represented – those of the elites? The disenfranchised? And what about non-human species (Sagoff 1988)? The practical problems involved with establishing such representation are apparent upon a moment's reflection (but see Weiss 1989 for

a proposed outline of an international system of legal obligations and duties which can support the implementation of intergenerational equity).

3. Both the complexity of the human and natural systems involved in industrial ecology and the need to deal with current uncertainty and emergent behavior as such systems evolve argue for the development of highly flexible legal tools. This is not trivial: because legal systems in many societies tend to be important components of social structure, they are usually conservative and relatively inflexible. Additionally, the inherent conservatism of legal structures is augmented by the tendency of regulation to create and nurture interests groups that benefit from it, and therefore come to constitute a significant barrier to subsequent regulatory rationalization. The price for this stability is paid in terms of inability to adjust to changing situations. Where change is rapid and fundamental, as it currently is with environmental issues, such inflexibility can lead to substantial inefficiency. Fortunately, there are a number of examples of such flexible mechanisms, ranging from schemes which, through emissions trading or similar policies, establish market systems designed to lead to efficient emissions reductions, to the 'covenant' system of The Netherlands, in which industry sectors and the government agree to binding, but flexible, contracts designed to reduce emissions to designated levels (Cairncross 1992; Matthews 1997; Biekard 1995; Netherlands Ministry of Housing, Spatial Planning and the Environment 1989, 1990, 1994a; OTA 1995).
4. Management of complex systems through flexible legal mechanisms imposes several requirements on the legal system if it is to be successful and stable over the long term. There must be adequate transparency to the policy development process: all stakeholders with a legitimate interest in the outcome should be represented as the regulations and implementation plans are developed (determining who has legitimate interests and how transparent the process should be will not necessarily be trivial in practice, and will probably be fact-dependent). There must also be performance validation mechanisms, such as deployment of sensor systems, data reporting requirements, or implementation of third party inspections. Finally, given the complexity of the natural and human systems involved, and the often considerable lag times involved in their dynamics, there must be mechanisms to assure that means and ends stay aligned over time. In most cases, these will probably take the form of long-term metrics or standards (Adriaanse 1993; Allenby 1999a).

The shift of environmental issues from overhead to strategic for firms and society as a whole require establishment of a more sophisticated environmental management system. Centralized command and control regulations will still be appropriate in some cases, especially where large-scale and irreversible impacts are possible: taking lead out of gasoline is an example; banning CFCs which cause stratospheric ozone depletion is another. In general, however, traditional environmental regulation is poorly suited for complex economic and technological systems. In such cases, establishing broader boundaries on behavior that motivates appropriate system evolution over time is much more effective. Product take back, which if properly implemented, internalizes to the producer the end-of-life costs associated with a product, is one example (Netherlands Ministry of Housing, Spatial Planning and the Environment 1994b). Regulations such as the 'community right-to-know' requirements in the USA, under which information regarding emissions is collected and submitted, and then made public by the regulator, are another.

More broadly, it is important to recognize that environmental regulators, or for that matter any centralized bureaucracy, become dysfunctional as the complexity of the system to be managed increases. To impose air scrubbers or water treatment requirements, for example, is a relatively simple matter: the technologies are not coupled to production and product design systems, and can be changed if inappropriate. On the other hand, once the jump is made in trying to regulate complex technological systems that are, in turn, coupled to other systems, the knowledge requirements and complexity of the systems involved increase beyond the ability of any central regulatory structure to manage. This is, in fact, why the economic structures of the Soviet Union and its satellites imploded. In such cases, a general rule would be, all else equal, that reliance on decentralized mechanisms such as the market is preferable to command and control approaches, and that regulatory management functions should be distributed so as to reflect the heterogeneity of the issues being addressed.

In this regard, it is also important to recognize the need to determine the appropriate jurisdictional boundaries. Political jurisdictions are creations of human culture and history, and there is no a priori reason why their boundaries should reflect underlying natural systems. It is thus no surprise that many problematic environmental perturbations are not coextensive with existing political boundaries. Europe, where virtually all riverine systems and airsheds are transboundary, is an obvious example, but by no means unique: emission of acid rain precursors in the USA or China cause acid rain in Canada or Japan; watershed degradation involving different nation states in Asia and the Middle East generates enormous legal and political conflict (Gleick 1993; US Department of State 1997). More subtly, industrial or consumer behavior may not be geographically or jurisdictionally co-located with the environmental perturbation to which it contributes. Thus, for example, much of the environmental impact of the economic activity of developed nation states is already embedded in the products or materials they import and, especially in the absence of prices which include all relevant social costs, will thus be virtually invisible to policy makers and consumers. Especially where the reach of the nation state does not extend – or, in some cases, cannot because of relevant international requirements such as trade law – this separation of behavior from impact can make management of such situations difficult. One obvious example of this mismatch in scale between political boundaries and environmental perturbations is Chernobyl, where a local power plant producing electricity for a national grid malfunctioned and created a European disaster (Shcherbak 1996).

Managing such issues inevitably requires complex negotiations, and probably always will. It is possible, however, to reduce the burden of such negotiations. For one thing, policies at each jurisdictional level, while addressing the specifics of the concern at that level, should reflect their impacts at all levels, and at the least not create unnecessary conflicts among levels. In particular, risk exportation to other jurisdictions is not a substitute for risk reduction, and should be avoided because it encourages the generation of externalities when the system is viewed as a whole. An obvious corollary is that harmonization of regulatory management structure at the same scale as the perturbation of concern is desirable. Thus, for example, the Montreal Protocol addressed the emission of ozone-depleting substances at the global scale, as did the Kyoto Protocol for global climate change. Given the behavior of the emitted substances and the global scale of the resultant impact, this is appropriate. On the other hand, a number of municipalities have passed regulations purporting to address the same phenomenon, in some cases adopting different standards, timelines and requirements than national or international agreements. Such 'symbolic legislation', which in many cases is not enforced in any event, is inappropriate. It not only is ineffectual in mitigating the perturbation, but generates substantial economic inefficiencies and, even if not enforced, results in inadvertent, sometimes virtually unavoidable, illegal behavior.

EXAMPLES OF SPECIFIC LEGAL ISSUES

On a social level, the transition of environmental issues from overhead to strategic inevitably implies conflict with existing legal and policy structures. Such structures – including, for example, those dealing with consumer protection, government procurement, antitrust, trade or national security – have generally been created over the years without any explicit consideration of their environmental implications. In effect, the environmental externalities associated with existing legal and policy regimes have been invisible to such institutions (Luhmann [1986] 1989). This is natural enough, given the treatment of such issues as overhead until recently. On the other hand, the broadening awareness of the fundamental linkages among cultural, technological, economic and environmental systems (Allenby and Richards 1994; Ayres and Simonis 1994; Socolow *et al.* 1994; Graedel and Allenby 1995; Grübler 1998; see also Chapter 39) has, at the same time, made the need to integrate environmental dimensions into existing legal systems more apparent. The environmental externalities imposed by existing legal and regulatory structures as they are currently constituted are seen as no longer acceptable. Several examples may clarify this transition, an important one for the industrial ecologist to understand.

Perhaps the example that springs first to mind is the conflict between trade and environment. Unlike many situations, here a core policy conflict is apparent. Trade policy as reflected in international agreements such as the North American Free Trade Agreement (NAFTA) and organizations such as the World Trade Organization (WTO), by and large seeks to facilitate the free transfer of goods and services among nation states. Environmental policy, on the other hand, seeks to control trade in environmentally unacceptable goods and, for some environmentalists, to impose developed-country standards restrictions on the means by which nation states produce goods and services internally. This means that, in at least some cases, real trade values actually are opposed to real environmental interests. For example, some European countries have imposed requirements that beverages be sold in returnable glass containers. The environmental purpose is to reduce the amount of waste produced by plastic or paper containers which are discarded, and to encourage the re-use of containers as opposed to the recycling of the material from which they are made (the degree to which the latter is environmentally preferable, and under what conditions, remains somewhat unclear). On the other hand, because of the weight of glass bottles, and the difficulty and expense of the reverse logistics system by which the bottles must be recovered and re-used, such a requirement clearly favors local (domestic) bottling operations and beverage producers such as brewers.

In many cases such as this one, not just regulations, statutes and treaties, but cultural models and world views are involved, and the synthesis of legal requirements is accordingly complicated by the need for the acculturation of, and mutual acceptance by, previously disparate groups. Thus, for example, a somewhat insular trade community that had heretofore dealt with environmental requirements, if it dealt with them at all, as protectionist trade barriers, is having to come to terms with environmentalists. The latter, in turn, tend to view the global economy, and thus trade, as suspect in itself, but an ideal tool to impose extraterritorial environmental requirements. Using trade in this way is, however, strongly constrained by international law, which significantly limits the ability to impose one country's environmental values on another through trade (Hartwell and Bergkamp 1994). Moreover, both groups are also beginning to understand that free trade, economic development and environmental protection are all valid policy goals, but it may not be possible to optimize all at the same time (Raul and Hagen 1993; Repetto 1995).

Several additional examples may illustrate both the dynamic of conflict leading to policy integration, and specific legal structures of interest to the industrial ecologist. Consider, for example, the structure of consumer protection law that many countries have implemented, which, in part, require that used products, or those containing used parts, be prominently labeled. The general purpose of these laws is to encourage full disclosure of the properties of the product by the vendor, and thus avoid fraud. Such a label, however, significantly reduces the price which can be charged for an article, and can hurt the trademark of the producer or vendor. On the other hand, re-use of products or parts can provide clear environmental benefits, and should thus be encouraged by public policy. Although this conflict has yet to be resolved, there are several obvious possibilities. As a stopgap measure in the short term, the principle could be established that, so long as a product, component or part meets all relevant specifications, it is immaterial whether it is used or not. In the longer term, the issue is one of consumer education: customers have been acculturated to avoid used products, or to value them less, and will need to be educated about the benefits of using used products. This process can be assisted by internalizing the positive externalities of such informed consumer choice – in short, by passing along the savings from using refurbished products and components to the consumer.

Another interesting example is government procurement regulations. This is a significant lever on producer behavior that has not been fully exploited. After all, governments have substantial buying power centralized in one organization, and thus can exercise significant control over a market (more technically, they can internalize costs that were previously externalities). To the extent that government procurement practices can be made environmentally preferable, therefore, they can exercise significant beneficial impacts on the performance of producers and vendors.

An important element of government procurement is the government standards and specifications, especially those associated with military procurement. These standards and specifications control a substantial amount of the design of many products, and the processes by which they are made, and, in many cases, predate any concern with the environment. They thus frequently embed environmentally problematic requirements within the economic system, and do so in a way that is invisible to most people. Thus, for example, the single biggest barrier to the US electronics industry's efforts to stop using CFCs, which were contributing to the breakdown of the stratospheric ozone layer, was military specifications and military standards (known as Milspec and Milstandard). Moreover, because of the tens of thousands of references to such requirements in myriads of procurement contracts and subcontracts, an enormous amount of work had to be done simply to change the welter of legal restrictions on using anything but CFCs (Morehouse 1994).

But perhaps the most interesting example is antitrust. As in the case of trade and environment, there are some fundamental issues regarding the relationship between antitrust and environmental policies. Antitrust seeks to maintain the competitiveness of markets by limiting the market power of firms, which generally means limiting their scope and scale (Nolan and Nolan-Haley 1990). Many environmental initiatives, such as post-consumer product take back, however, seek to do the exact opposite: to expand the scope and scale of the firm so that it is responsible for the environmental impact of its product from material selection through consumer use to take back, and recycling or refurbishment. The one seeks an atomistic market with no central control; the other seeks to extend the control of firms in the interest of internalizing to them the costs of negative environmental externalities (and benefits of positive externalities).

The dichotomy between antitrust and environmental policies is exacerbated by the question of technological evolution. By and large, technological evolution is most rapid in competitive markets with low barriers to the introduction of new technologies. Such market structures are likely to be fostered by traditional antitrust policies. On the other hand, if firms are to implement environmentally preferable practices across the life cycle of their product, they will generally have to develop a means of linking the technologies used at various points in the product life cycle. Thus, for example, the technologies used to disassemble the product after the consumer is through with it need to be considered in the initial design of the product (a process called by designers, reasonably enough, 'Design for Disassembly'). Linking technologies in such a way creates a more complex, coevolved, technological system and reduces the ability to evolve any part of that system rapidly.

Thus, on the one hand, industrial ecology indicates that rapid evolution of environmentally and economically more efficient technologies is critical to moving towards sustainability in the short term, but, on the other hand, it encourages the development of systems which reduce the potential for such evolution. The solution to this dilemma – to understand which structure is economically and environmentally better under what conditions – requires an analytical sophistication that does not yet exist. It is an indication of the legal challenges which industrial ecology both raises and must address.

CONCLUSION

The legal context within which industrial ecologists operate is complex and varies by jurisdiction. Moreover, once it is recognized that environment is increasingly strategic, rather than overhead, for firms and for society as a whole, it follows that the traditional discipline of environmental law is less and less important for the practice of industrial ecology. Rather, the industrial ecologist needs to become broadly familiar with the legal structures and issues which affect development, trade, and economic and technology policy, and comfortable in a global governance system that is both unclear and evolving rapidly.

7. Industrial ecology and industrial metabolism:use and misuse of metaphors

Allan Johansson

THE VALUE OF METAPHORS

The use of visual metaphors goes far back in human history. Early evolutionary evidence indicates that, about 35000 years ago, humans began to use body ornaments that evoked qualities of animal species (Seitz 2000; White 1989). They also sculpted abstract designs that are believed to be depictions of objects and represent the transfer of patterns in nature to a context in which they function aesthetically, that is to say as visual metaphors.

Metaphors according to Aristotle 'are a device that consists in giving the thing a name that belongs to something else' (Eisenberg 1992). But their use goes much deeper than that; they constitute an important instrument for transferring meaning, in correspondence with the original Greek derivation 'metapherein', to transfer (Seitz 2000). By giving a new name to something one implicitly, but discretely, conveys the thought that some, but not all, of the characteristic properties are carried over, together with the name. It is this element of 'wishful thinking' that causes problems in the use of metaphors in science. Predominantly an artistic instrument, the use of metaphors involves a certain poetic indeterminacy. This allows for a new dimension of communication through the play of imagination, which goes beyond the possibilities of formal strict verbal communication. In literature the theories of metaphors have generally concentrated on studies of their use in language and literature only, and it is only recently that more systematic studies have been devoted to their importance in not only conveying messages but also in shaping thought (Johnson-Sheehan 1997).

In the latter context it appears, perhaps somewhat surprisingly, that the use of metaphors has been particularly frequent and fruitful in natural sciences, despite the fact that precision and exactness of expression, not poetic qualities, are generally the qualifications linked to scientific communication. However, recent research does in fact indicate that metaphors play an important part in the brain's cognitive mechanisms by involving the perception between disjoint domains of experience. It is this perceived relationship between these domains that we represent in different symbol systems as metaphorical (Seitz 2000). Thus the concept extends far beyond its use as an artistic tool for poetic language, into the realm of creative thinking, where it functions as an inducer of new ideas.

This extension of creative thinking is not neutral, however. It is often a conscious, although not always openly declared, effort to account for controversial value questions that cannot be dealt with by pure logic. Early uses of metaphors in science often relate to religious symbolism. They can be seen as efforts to resolve the feeling of the growing moral conflict between man's unlimited quest for more knowledge and God's supremacy and essential mystery.

Examples of the creative impact of metaphors in science are numerous, albeit often disputed or refuted as later dramatization (Johnson-Sheehan 1997). Some of the more often cited are those of Darwin's living tree metaphor for the evolutionary process (Gruber 1978), Kekulé's snake image (which, in a dream, allegedly gave him the idea of the structure of benzene) and the structure of DNA as a spiral staircase. Szilard got the idea for a sustainable nuclear reaction by watching traffic lights turn from red to green. Many of Einstein's creative ideas, too, apparently derived from visual experience in which spoken or written language played no substantive role (Dreistadt 1968).

To be sure, natural science has its own language, mathematics. Purists may think that it is the only language acceptable when describing natural events. Yet it turns out that very complex relations, so common in biology, social science and economics, are difficult or impossible to describe properly in mathematical terms. Even on a rather trivial level the intricate systemic interactions become unmanageable. Thus new tools must be invoked for the communication of ideas.

In such cases the metaphoric tools are not only a means for describing an idea. They become fundamental parts of the understanding itself. Thus, paraphrasing Winston Churchill's famous comment about democracy, metaphors may be misleading, but they are the least misleading thing we have. We use metaphors to improve our communication, often by extending our thinking into the domains of intuition by invoking an imaginary example of something our interlocutor(s) or audience are supposed to be familiar with.

Correctly used, metaphors can actually contribute to important extensions of human understanding of nature. Sometimes metaphoric examples from other disciplines of science can work fruitfully when adopted in different domains. A classical example, and perhaps one of the most successful, is provided by Sadi Carnot, the young French engineer/physicist, who used the metaphor of water running from a higher altitude to a lower, when formulating his ideas on how useful work could be extracted from heat. In his metaphor an unknown heat substance 'caloric' flows like water from a higher temperature to a lower, making it possible to extract useful work from this dynamic flow as work can be extracted from running water.

In spite of the fact that Carnot's basic assumption of 'caloric' was wrong, the metaphor worked so well that he was able to formulate a fundamental theorem of thermodynamics that describes the conditions and maximum efficiency of such a fundamental piece of engineering as a heat engine, without knowing its detailed construction. Carnot's theorem is still used today as a design tool for sophisticated energy-based systems, like power plants and combined cogeneration systems. Further, it constitutes one of the few examples where a solution to an engineering problem has actually provided new physical insight (Feynman *et al.* 1963).

THE USE OF METAPHORS IN ENVIRONMENTAL SCIENCES

Metaphors allow us to carry our thoughts out into the unknown by using the known as a stepping stone and their influence is clear when they are explicitly used to describe an issue. This influence becomes less evident and more subtle and often also more persuasive when they become so deeply attached to a subject that the participants no longer are aware of its being a metaphor.

Metaphors are particularly tempting to use in the area of environmental work, owing to its nature of interactions between complex systems, and ethics, which are impossible to describe in exact quantitative terms. As the central issue in environmental protection is the interrelation between the man-made, anthropogenic activities and those of the natural environment, it is not surprising to find that this is also reflected by the metaphors used. We speak commonly of eco-efficiency, eco-industry, ecodesign, ecorestructuring, and so on.

But metaphors have cultural and professional limitations. Their use assumes that the interlocutor is familiar with the example and has the same perception of it. The latter is particularly relevant when we consider the earlier mentioned question of values frequently linked to the metaphoric expressions. It so happens that these conditions are frequently not met. This is particularly problematic in the complex world of environmental protection or management, which is loaded with emotional baggage and cultural heritage. In such a case there is the obvious risk that the metaphor, if persuasive enough, is given formal validity and starts to live a life of its own. It is taken too literally and used to extend the original idea beyond its limits. When this happens a potentially fruitful support for creative thinking not only loses its potency, it actually works against its real purpose of communicating understanding, and provokes confusion instead. The issue of overextending or misusing metaphors is not new, and by no means limited only to the environmental field. It has, in a broader context, as pointed out by Eisenberg (1992), been dealt with by philosophers including Hobbes,[1] Locke[2] and Wittgenstein.[3] More recently, Sokal (1997) has complained about the intellectual dishonesty of using metaphors to invoke scientific precision where there is none.[4] He properly insists that the purpose of a metaphor is to elucidate a new and unfamiliar concept by taking support from a familiar one, not the converse.

Recently, a whole range of ecologically related new sub-disciplines has emerged, and it is not always easy to see when the metaphor used as a name of a discipline actually adds insight and when it only conveys wishful thinking. Perhaps owing to the nature of the issue, but also occasionally owing to concealed ambitions, this is an area where the hidden message of the metaphor may be particularly deceiving. The metaphors often express non-scientific sentiments, such as fear, threat or desirability, which the author chooses to link to the issue. In the field of environmental science these sentiments often revolve around the central moral dilemma of Man's relation to, and responsibility for, Nature. This situation is in fact similar to the early use of metaphors in an effort to unite science and religion. 'Mother' nature is one such metaphor. Lovelock's 'Gaia' metaphor is a more recent and more deliberate example of the double-edged nature of metaphors in science (Lovelock 1972, 1979, 1988). There is reason to believe that it is the divine nature of the 'Gaia' concept that has penetrated into the mind of the greater public, not its scientific content. On a slightly different level, the terms eco-efficiency, eco-industry, ecodesign and others with the eco-prefix are perceived as 'good' by virtue of the name only, without further scrutiny of the content. [*Ed. note (RUA)*: I was told by a participant that the committee planning the well-known 1992 summer study, which firmly established the phrase 'industrial ecology', had rejected the term 'industrial metabolism' because it might remind people of indelicate biological functions, such as excretion, whereas ecology has 'good' associations.] The current trend of increasing pressure to 'sell' science also functions as an invitation to invent new suggestive names for essentially old activities, simply to break through barriers in political and economic decision making (and funding).

THE HIDDEN MESSAGE OF METAPHORS IN ENVIRONMENTAL SCIENCE

In order to illustrate the sensitivity of the issue, the remainder of this chapter attempts to analyze the real meaning, and some more or less commonly accepted implicit extensions, of three rather well-established and closely related concepts: *ecosystem management*, *industrial metabolism* and *industrial ecology*.

Ecosystem management expresses the notion of including the natural surroundings into our planning, as most human activities have an impact on the surrounding ecosystem (see, for example, Christensen *et al.* 1996). Thus ecosystem management can be seen as a neutral, simple and straightforward extension of a rational resource management effort. But there is also a larger, partially concealed, implicit message. When we speak of ecosystem management, we not only convey the need for proper attention to the ecosystem. The word 'management' also implies the need for a businesslike, utilitarian management approach putting human needs in the center. The ecosystem is thus presented as something to be managed – and hence utilized – without ever bringing the issue explicitly to the table. An important element of today's debate around sustainability is consequently neglected altogether.

Industrial metabolism conveys the descriptive idea of the industrial system as a living complex organism, 'feeding' on natural resources, material and energy, 'digesting' them into useful products and 'excreting' waste. This is a rather value-neutral description helping us to see the need for a broader view, focusing on interactions of material and energy flows, rather than on single issues as previously was the case. In passing, it is interesting to note that, while industrial metabolism metaphorically suggests that machines behave like living cells, an early metaphor used by Descartes (one of the architects of the Enlightenment) used the same metaphor 'nature is a machine' to increase his understanding of nature.[5]

Industrial metabolism traces material and energy flows from initial extraction of resources through industrial and consumer systems to the final disposal of wastes. It makes explicit use of the mass balance principle. First developed by Ayres and collaborators in a series of papers and books (Ayres *et al.* 1989; Ayres 1989a, 1989b, 1993b, 1994b; Ayres and Simonis 1994) industrial metabolism has become an important foundation of industrial ecology. Industrial metabolism can usefully be applied at many different levels: globally, nationally, regionally, by industry, by company and by site. By invoking the parallel to biological metabolism, industrial metabolism analysis highlights the dramatic difference between natural and industrial metabolic processes, in particular the large difference in energy and material densities and fluxes and the lack of a primary producer (analogous to photosynthetic organisms) in the industrial world. Also, in natural systems, some nutrients flow in closed loops with near universal recycling, whereas industrial systems are mostly dissipative, leading to materials concentrations too low to be worth recovering but high enough to pollute.

So far industrial metabolism studies have tended to focus on flows of chemicals and metals, but the approach is also useful in analysis of energy and water flows. Some companies have conducted environmental audits based on this method and regional application gives valuable insight into the sustainability of industry in natural units such as watersheds or atmospheric basins. Mapping sources, processes and transformations, and sinks in a region, offer a systemic basis for public and corporate action. In an early application, Ayres *et al.* (Ayres and Rod 1986; Ayres *et al.* 1988) studied the historical development of pollutant levels in the Hudson–Raritan basin over the period 1880–1980. A similar study has also been made for tracing chromium and lead poisoning in Sweden over the period 1880–1980 (Lohm *et al.* 1994). The International Institute for Applied System Analysis (Stigliani, Jaffé and Anderberg 1993) has completed an industrial metabolism study of the Rhine basin, the most ambitious application so far. The study examined for the whole basin sources of pollution and pathways by which pollutants end up in the river. Materials studied include cadmium, lead, zinc, lindane, PCBs, nitrogen and phosphorus. The results suggest that, in the Rhine basin, industry has made major progress on reducing emissions. However, there are increasing flows of pollution from 'non-point' or diffuse sources, including farms, consumers, runoff from roads and highways, and disposal sites. The findings are of great value in design of policy, industrial practice and public education. Experience seems to suggest that industrial metabolism, in spite of its suggestive biological symbolism, can be used as a practical tool for further understanding of the complex relations of material and energy fluxes in the industrial context.

Industrial ecology is a concept whose origins are not so easy to trace (but see Erkman 1998; Chapter 9). The concept existed in the 1970s, well before the name became popular (Watanabe 1972; Odum 1955; Hall 1975). In those times it was often used almost synonymously with industrial metabolism. However, as now understood, industrial ecology goes further than industrial metabolism: it attempts to understand how the industrial system works as an interactive system, how it can be regulated and how it interacts with the biosphere and other industrial systems. The latter is an important extension. But the crucial question arises: is industrial ecology only a descriptive name: is it a tempting vision? Or can it be used as an actual guide for industrial planning in an effort to mimic ecological dependencies in nature? If the latter holds true, important implications could be developed to provide guidance to restructure the industry to make it more compatible with its natural environment.

Industrial ecology is claimed to be 'ecological' in that it places human activity 'industry' in the very broadest sense in a larger context of the biophysical environment from which resources are extracted and which is negatively affected by the emissions of effluents and wastes (Lifset 1997). It is also claimed that the natural systems can function as models for the man-made system in terms of efficient use of resources, energy and wastes. In this context, the famous Kalundborg example is often cited (for example, Ehrenfeld 1997). (Kalundborg is an industrial city in Denmark, where wastes from petroleum refineries and a power plant are profitably utilized in other industries.) On the critical side, voices have been raised that what has been achieved in Kalundborg is in fact standard engineering practice realized in many other places as well (Johansson 1997).

Thus, when we speak of industrial ecology, we want to further broaden the picture to include the possible interaction of different industrial systems with each other, and their future development in relation to each other, drawing upon our knowledge from natural systems (Ayres, Rod and McMichael 1987; Ayres *et al*. 1989; Ausubel and Sladovich 1989; Patel 1992; Allenby and Richards 1994; Richards and Frosch 1994; Schultze 1996). In some applications the borderline between science and wishful thinking becomes very thin. Further, in the author's view, it remains yet to be proved that this particular metaphor actually can be helpful in defining new strategies for industrial development. Natural processes can only evolve by reacting to changes. However, anthropogenic systems should be able to foresee both risks and opportunities, and actively plan ahead.

In conclusion, one could say that successful metaphors in science are often victims of their own success. The imaginative strength of a successful metaphor takes over, migrates to other domains in science, and carries the message further than was intended, frequently conveying feelings and values that are not justified by the scientific content. This can have particularly far-reaching consequences in the formulation of environmental policies, an area where important social and economic decisions are made by non-scientists. But we must keep in mind that this indeterminacy is organically linked to the use of metaphors: the extension of the imagination beyond what is known as truth is their very reason for existence, it is a tool for bringing creative intuition into science, and occasionally also to a greater public.

Reverting to Carnot's case, mentioned earlier, there was no such substance as 'caloric'. In fact, it is the incoherent thermal motion of atoms or molecules that is partly converted to directed motion in a heat engine. Nevertheless, the metaphor worked and laid the foundation of a new branch of physics–thermodynamics. It also contributed to a wonderful step forward in engineering and, indirectly, to the 'age of steam' and the 'industrial revolution' (a metaphor, of course).

NOTES

1. In *Leviathan*: 'When [men] use words metaphorically; that is, in other sense than they are ordained for, [they] thereby deceive others. Such [inconstant] names can never be true grounds for any ratiocination' (Hobbes 1982 [1651]).
2. In *An Essay concerning Human Understanding*: 'Figurative applications of words . . . are for nothing else but to insinuate wrong ideas, move the passions, and thereby mislead the judgement, and so indeed are perfect cheats. They are certainly, in all discourses that pretend to inform or instruct, wholly to be avoided' (Locke 1998 [1689]).
3. In *Tractatus Logico-Philosophicus*: 'Of what we cannot speak we must remain silent' (Wittgenstein 1995 [1921]).
4. In *La Recherche*: 'Le rôle d'une métaphore est d'éclairer une idée peu familière en la reliant à une autre qui l'est plus, pas l'inverse.'
5. In *Principia Philosophiae*: 'And I have been greatly helped by considering machines. The only difference I can see between machines and natural objects is that the workings of machines are mostly carried out by apparatus large enough to be readily perceptible by the senses . . . whereas natural processes almost always depend on parts so small that they utterly elude our senses' (Descartes 1969 [1644]).

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PART II

Methodology

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8. Material flow analysis

Stefan Bringezu and Yuichi Moriguchi

Understanding the structure and functioning of the industrial or societal metabolism is at the core of industrial ecology (Ayres 1989a; see also Chapters 1, 2 and 3). Material flow analysis (MFA) refers to the analysis of the throughput of process chains comprising extraction or harvest, chemical transformation, manufacturing, consumption, recycling and disposal of materials. It is based on accounts in physical units (usually in terms of tons) quantifying the inputs and outputs of those processes. The subjects of the accounting are chemically defined substances (for example, carbon or carbon dioxide) on the one hand and natural or technical compounds or 'bulk' materials (for example, coal, wood) on the other hand. MFA has often been used as a synonym for material flow accounting; in a strict sense the accounting represents only one of several steps of the analysis, and has a clear linkage to economic accounting.

MFA has become a fast-growing field of research with increasing policy relevance. All studies are based on the common paradigm of industrial metabolism and use the methodological principle of mass balancing. However, there are various methodological approaches which are based on different goals, concepts and target questions, although each study may claim to contribute to knowledge of the industrial metabolism. In 1996, the network ConAccount was established to provide a platform for information exchange on MFA (*www.conaccount.net*). A first inventory on MFA projects and activities was provided (Bringezu *et al.* 1998a). Several meetings took place (Bringezu *et al.* 1997, 1998b; Kleijn *et al.* 1999) and a research and development agenda was defined through an interactive process (Bringezu *et al.* 1998c).

The diversity of MFA approaches derives from different conceptual backgrounds. The basic concept common to many studies is that the industrial system together with its societal interactions is embedded in the biogeosphere system, thus being dependent upon factors critical for the coexistence of both systems (Ayres and Simonis 1994; Baccini and Brunner 1991, see also Chapter 2). The paradigm vision of a sustainable industrial system is characterized by minimized and consistent physical exchanges between human society and the environment, with the internal material loops being driven by renewable energy flows (for example, Richards *et al.* 1994). However, different strategies have been pursued to develop industrial metabolism in a sustainable fashion.

One basic strategy may be described as *detoxification* of the industrial metabolism. This refers to the mitigation of the releases of critical substances to the environment by pollution reduction. In a wider sense, this relates to any specific environmental impact such as toxicity to human beings and other organisms, eutrophication, acidification, ozone depletion, global warming and so on. Regulatory governmental actions in terms of substance bans and restrictions of use represented the first measures of environmental policy (see Chapter 6). The concept of cleaner technology is aimed primarily towards the mitigation of critical releases to the environment (see Chapter 4). It is possible that, as a consequence of the effectiveness of such measures, pollution problems in the spatial–temporal short range could be solved. Transregional and global problems and problem shifting to future generations, however, as well as the complexity of the industrial metabolism, made it necessary to analyze the flows of hazardous substances, selected materials or products in a systems-wide approach; that is, from cradle to grave, and with respect to the interlinkage of different flows.

Another complementary strategy may be regarded as *dematerialization* of the industrial metabolism. Considering the current quantity of primary resource use by industrial economies, an increase of resource efficiency by a factor of 4 to 10 was proposed (Schmidt-Bleek 1994a, 1994b; Weizsäcker *et al.* 1997). This goal has been adopted by a variety of international organizations and national governments. On the program level the factor 4/10 concept was adopted by the special session of the United Nations (UNGASS 1997) and the World Business Council for Sustainable Development (WBCSD 1998). The environmental ministers of the OECD (1996a) urged progress towards this end. Several countries included the aim in political programs (for example, Austria, the Netherlands, Finland and Sweden; see also Gardener and Sampat 1998). In Scandinavian countries research was launched to test the broad-scale feasibility of factor 4/10 (Nordic Council of Ministers 1999). In Germany a draft for an environmental policy program (BMU 1998) refers to a factor of a 2.5 increase in productivity of non-renewable raw materials (1993 to 2020). An increase in eco-efficiency is now considered essential by the environmental ministers of the European Union (1999). The review of the Fifth (environmental) Action Programme (Decision No 2179/98/EC) emphasizes resource use and efficiency.

The factor concept aims at the provision of increased services and value-added with reduced resource requirements. Dematerialization of the economy may imply a diminution of all hardware products and thus the throughput of the economy as a whole, comprising the use of primary *and* secondary materials. However, dematerialization may also be directed more specifically to the reduction of the primary inputs and/or final waste disposal. The concept of eco-efficiency includes not only the major inputs (materials, energy, water, area) but also the major outputs to the environment (emissions to air, water, waste) and relates them to the products, services or benefits produced (EEA 1999a; OECD 1998b; Verfaillie and Bidwell 2000). However, for the environment the reduction of the absolute impacts through material flows is essential. Thus, the quantity of human-induced material flows through the industrial system must also be adjusted to adequate levels of exchange between the economy and the environment.

TYPES OF ANALYSIS

In the above context, two basic types of material flow-related analyses may be distinguished according to their primary focus; although in practice a continuum of different approaches exists (Table 8.1). Neither type I nor type II is strictly coincident with the above-mentioned two paradigmatic strategies. However, the importance of the detoxification concept seems highest in Ia and lowest in IIc. In contrast, the intention to support dematerialization seems highest in analyses of IIc and lowest in Ia. Nevertheless both complementary strategies are increasingly being combined, especially in Ic and IIa. Whereas type I analyses are often performed from a technical engineering perspective, type II analyses are more directed to socioeconomic relationships.

Table 8.1 Types of material flow-related analysis

Type of analysis	I		
	a	b	c
Objects of primary interest	Specific environmental problems related to certain impacts per unit flow of:		
	substances e.g. Cd, Cl, Pb, Zn, Hg, N, P, C, CO_2, CFC	materials e.g. wooden products, energy carriers, excavation, biomass, plastics	products e.g. diapers, batteries, cars
	within certain firms, sectors, regions		
	II		
	a	b	c
	Problems of environmental concern related to the throughput of:		
	firms e.g. single plants, medium and large companies	sectors e.g. production sectors, chemical industry, construction	regions e.g. total or main throughput, mass flow balance, total material requirement
	associated with substances, materials, products		

Source: Adapted from Bringezu and Kleijn (1997).

Type Ia

Substance flow analysis (SFA) has been used to determine the main entrance routes to the environment, the processes associated with these emissions, the stocks and flows within the industrial system as well as the trans-media flows, chemical, physical, biological transformations and resulting concentrations in the environment (see Chapter 9). Spaciotemporal distribution is of high concern in SFA. Results from these analyses are often used as inputs to further analyses for quantitatively assessing risks to substance-specific endpoints.

A variety of studies have been conducted on toxic heavy metals such as arsenic, cadmium, chromium, copper, mercury, lead and zinc (Ayres, Ayres and Tarr 1994; Ayres and Ayres 1996; Ayres and Ayres 1999a; Reiner *et al.* 1997; Dahlbo and Assmuth 1997; Maag *et al.* 1997; Hansen 1997; Maxson and Vonkeman 1996; Voet *et al.* 1994; see also Chapters 27 and 28).

Nutrients such as nitrogen and phosphorus are taken into account mainly because of eutrophication problems and the search for effective mitigation measures (Ayres and Ayres 1996; Voet 1996).

The flow of carbon is studied because it is linked to global warming due to current fossil fuel dependence. The accounting for carbon dioxide and other greenhouse gas emissions and the study of trends, sources, responsible technologies, possible sinks and measures for abatement have been increasingly reported by statistical services.

The flow of chlorine and chlorinated substances has been subject to various studies owing to the toxic potential and various pollution problems through chlorinated solvents and persistent organochlorines (Ayres and Ayres 1999a; Kleijn *et al.* 1997), the ozone-depleting effect of CFCs (Obernosterer and Brunner 1997) and a controversial debate over risks incurred through incineration of materials such as PVC (Tukker 1998).

Type Ib

Selected bulk material flows have been studied for various reasons. Resource extraction by mining and quarrying was studied to assess the geomorphic and hydrological changes due to urbanization (see Chapter 28). The flow of biomass from human production has been studied to relate it to biomass production in natural ecosystems in order to evaluate the pressure on species diversity (Vitousek *et al.* 1986; Haberl 1997).

On the one hand, metals like aluminum, timber products like pulp and paper, and construction aggregates represent important base materials for industrial purposes. On the other hand these flows – although per se rather harmless – may be *linked* with other flows significantly burdening the environment, for example, the 'red mud' problem with alumina production and the energy-intensive production of aluminum (Ayres and Ayres 1996). Base materials such as plastics have been subject to various studies on the potentials and environmental consequences of recycling and cascading use (for example, Fehringer and Brunner 1997; Patel 1999).

Possible effects of alternative technologies and materials management on global warming potential have been studied, for example for construction materials (Gielen 1999). This kind of analysis is related to studies of types Ic and IIb.

Type Ic

When the environmental impacts of certain products and services is the primary interest, the approach is normally denoted life cycle assessment (LCA). The product LCA literature is reviewed in Chapter 12. In general, the system boundary of LCA ('cradle to grave') corresponds with the systems perspective of the anthroposphere, technosphere or physical economy. Some methods of evaluation may be used for LCA and MFA as well (see Chapter 13).

From type Ia to type Ic the primary interest becomes increasingly comprehensive and complex (Table 8.1). It commences with the analysis of selected substances, considered compound materials and progresses to products consisting of several materials. Not only the number of potential objects but also the number of potential impacts per study object increases by several orders of magnitude. The complexity of the associated chain net also grows.

Type IIa

The primary interest may lie in the metabolic performance of a firm or household, a sector or a region. In this case, there may be no or insufficient information about specific environmental problems. Often the main task is to evaluate the throughput of those entities in order to find the major problems, support priority setting, check the possibilities for improvement measures and provide tools for monitoring their effectiveness.

Accounting for the physical throughput of a firm is becoming more and more commonplace, at least for bigger companies. It is found in corporate environmental reporting. Materials accounts are used for environmental management (see Orbach and Liedtke 1998 for a review for Germany). Eco-efficiency at the firm level has been indicated in reports (for example, WBCSD 1998, 1999 – method overview and pilot study results; Verfaillie and Bidwell 2000 – program activities). Flow analyses of materials have been applied for optimization within companies (Spengler 1998). However, the limited scope of firm accounts calls for complementary analyses with a wider systems perspective, either through LCA-type analyses for infrastructures (Bringezu *et al.* 1996) and main products (for example, Liedtke *et al.* 1998) or by analyses of higher aggregates of production and consumption, that is analyses of total production sectors or whole economies.

Type IIb

When the primary interest is devoted to certain industrial sectors or fields of activity, MFA may be used to identify the most critical fluxes in terms of quality and/or quantity. For instance, different industrial sectors may be compared with regard to various inputs and outputs either from other sectors or from the environment (Ayres and Ayres 1998; Hohmeyer *et al.* 1997; Windsperger *et al.* 1997). When the analysis comprises all sectors of a region or national economy in a comparative manner, the accounting is closely related to type IIc; in that case the main interest may still be devoted to the national economy as a whole and the sectoral analysis serves to indicate those sectors which are of prior importance regarding criteria of specific interest (for example, CO_2 emission intensity or resource intensity). In those cases, a top-down approach is usually applied. Certain sectors or activities may be analyzed in detail, for example, the construction sector (Glenck and Lahner 1997; Schandl and Hüttler 1997) or activities such as nutrition, cleaning, maintaining a dwelling and working, transport and communication (Baccini and Brunner 1991). Analyses of this type may have strong interrelations to type Ib, as when for instance construction material flows are accounted for in a comprehensive manner (Bringezu and Schütz 1998; Kohler *et al.* 1999).

Type IIc

A major field of MFA represents the analysis of the metabolism of cities, regions and national or supranational economies. The accounting may be directed to selected substances and materials or to total material input, output and throughput.

The metabolism of cities was analyzed in early studies by Wolman (1965) and Duvigneaud and Denayer-DeSmet (1977) and thoroughly for the case of Hong Kong (Boyden 1980; Koenig 1997) and Vienna (Obernosterer *et al.* 1998). For a review, see Einig (1998). At the regional level a comprehensive milestone study was performed by Brunner *et al.* (1994) for the Swiss valley, Bünztal. The flow of pollutants was analyzed by Stigliani and Anderberg (1994) for the Rhine basin. The metabolism of the old industrialized German Ruhr region was studied by Bringezu and Schütz (1996b). Economy-wide MFA at the national level has attracted special attention (see below). The main interest lies in the overall characterization of the metabolic performance of the studied entities, in order to understand the volume, structure and quality of the throughput and to assess the status and trend with regard to sustainability.

The term 'MFA' has usually referred to analyses of types Ia, Ib, IIb and IIc. Studies of type Ic are generally considered to fall under the heading of LCA. Accounting of type IIa is mainly related to environmental management. There are also combinations of regional and product-oriented analyses. Accounting for the hidden flows of imports (and exports), that is upstream resource requirements of imported (or exported) products, may be combined with the domestic resource requirements of a regional or national economy in order to provide the total material requirements (TMR) (and total material consumption – TMC) indicators (Bringezu *et al.* 1994; Adriaanse *et al.* 1997). Nevertheless, all of these analyses use the accounting of material inputs and outputs of processes in a quantitative manner, and many of them apply a systems or chain perspective.

USE OF MATERIAL FLOW-RELATED ANALYSES

In general MFA provides a system-analytical view of various interlinked processes and flows to support the strategic and priority-oriented design of management measures. In line with environmental protection policy as it has evolved since the 1960s, type Ia analyses have been applied to control the flow of hazardous substances. The results contributed to public policy in different ways (Bovenkerk 1998; Hansen 1998):

- The analyses assisted in finding a consensus on the data which is an important prerequisite for policy measures.
- MFA has led to new insights and to changes in environmental policy (for example, abandoning the aim of closed chlorine cycling in favor of controlling the most hazardous emissions).
- The analyses discovered new problems (for example, the mercury stocks in chlorine plants).
- They also contributed to finding new solutions (for example, source-oriented input reduction in the case of non-degradable substances).

The use and policy relevance of type II analyses have been increased in recent years in the following ways (Bringezu 2000b):

- support for policy debate on goals and targets, especially with regard to the resource and eco-efficiency debate and the integration of environmental and economic policies,
- number of companies providing firm and product accounts,
- provision of economy-wide material flow accounts for regular use in official statistical compilations,
- derivation of indicators for progress towards sustainability.

PROCEDURE AND ELEMENTS OF THE ANALYSIS

Although there is no general consensus on a methodological framework for materials accounting and flow analysis, the procedure and some elements of the studies have essential features in common (see reports of the ConAccount focus groups 'Towards a general framework for MFA' in Bringezu *et al.* 1997, pp. 309–22). The procedure usually comprises four steps: goal and systems definition, process chain analysis, accounting and balancing, modeling and evaluation.

The *systems definition* comprises the formulation of the target questions, the definition of scope and systems boundary. *Target questions* are defined according to the primary objectives. In all types of analysis, it has to be determined which flow categories will have to be accounted in order to quantify volume and path of the flows, and to find out those flows which are most relevant and crucial for the problems of primary interest, and those factors most responsible for these flows. The *scope* defines the spatial, temporal and sometimes functional extent of the studied objects. The categorized flows are studied along their path that is related to spatially defined compartments or regions or to functionally defined industrial sectors. The flows are always accounted on the basis of a temporally defined period. The scope may be similar for type I and type II. The system boundary defines the start and the end of the material flows which are accounted. It is – at least – partly determined by the scope but may comprise additional functional elements, such as the borderline between the environment and the economic sectors of a region. Scope and system boundary are not necessarily identical, especially when regionally oriented accounts are combined with product chain-oriented accounts. For instance, if the transnational material requirement of a national economy is determined (for example, as part of TMR, see below) the scope remains national while the system boundary is defined functionally on a larger scale.

At a certain level of detail the *process chain analysis* defines the processes for which the inputs and outputs are to be determined quantitatively by *accounting and balancing*. Here the fundamental principle of mass conservation is used to balance inputs and outputs of processes and (sub)systems (Ayres and Ayres 1999a). The balancing is used to check accuracy of empirical data, to improve consistency and to 'fill in' missing data. This is usually performed on the basis of stoichiometric or technical coefficients (for example, Windsperger *et al.* 1997; Bringezu *et al.* 1998a) and may be assisted by computer simulation (Ayres and Ayres 1999a), based on mathematical modeling (Baccini and Bader 1996).

Modeling may be applied in the basic form of 'bookkeeping' or with increasing complexity as static and dynamic modeling (see Chapter 9). The *evaluation* of the results is related to the primary interest and basic assumptions. The criteria may focus on the indication of (a) specifically known impacts per unit of flow. Here impact coefficients can be applied, for example for ozone depletion (see Chapter 13). The criteria may (b) indicate a generic environmental pressure potential. In this case, the volume[1] of flows (for example, water consumption, materials extraction) may be used to monitor certain pressures over time. More elaborate, but still generic, criteria can be based on energy flow-based parameters such as exergy (Ayres and Ayres 1999a) or emergy (Odum 1996).

ECONOMY-WIDE MFA

Material flow accounts may quantify the physical exchange of national economies with the environment. After the first approaches of Ayres and Kneese (1969), domestic MFAs were established independently for Austria (Steurer 1992), Japan (Japanese Environmental Agency 1992) and Germany (Schütz and Bringezu 1993). Aggregated material flow balances comprise domestic resource extraction and imports (inputs) and domestic releases to the environment and exports (outputs), as shown in Table 8.2. Upstream or downstream flows associated with imports and exports (resource requirements or emissions) may also be taken into account. A sectoral disaggregation can be provided by physical input–output tables (see Chapter 10).

Table 8.2 Economy-wide material balance with derived indicators

Inputs (origin)	Outputs (destination)
Domestic extraction	Emissions and wastes
Fossil fuels (coal, oil etc.)	Emissions to air
Minerals (ores, gravel etc.)	Waste landfilled
Biomass (timber, cereals etc.)	Emissions to water
	Dissipative use of products
Imports	(Fertilizer, manure, compost, seeds etc.)
Direct material input (DMI)	*Domestic processed output to nature (DPO)*
Unused domestic extraction	Disposal of unused domestic extraction
From mining/quarrying	From mining/quarrying
From biomass harvest	From biomass harvest
Soil excavation	Soil excavation
Total material input (TMI)	*Total domestic output to nature (TDO)*
	Exports
	Total material output (TMO)
	Net additions to stock (NAS)
	Infrastructures and buildings
Upstream flows associated with imports	Other (machinery, durable goods etc.)
Total material requirements (TMR)	Upstream flows associated with exports

Note: Excludes water and air flows (unless contained in other materials).

Source: Adapted from Eurostat (2000).

MFA has also entered official statistical compendia within the framework of integrated environmental and economic accounting (Radermacher and Stahmer 1998; see also Chapter 14). A methodological guide has been prepared by Eurostat (2000). National material accounts exist for Austria (Schandl 1998; Gerhold and Petrovic 2000; Matthews *et al*. 2000), Denmark (Pedersen 1999), Germany (see Chapter 23), Finland (Muukkonen 2000; Statistics Finland 1999; Mäenpää *et al*. 2000), Italy (De Marco *et al*. 1999; Femia 2000), Japan (see Chapter 24), the Netherlands (Matthews *et al*. 2000), Sweden (Isacsson *et al*. 2000), the UK (Vaze and Barron 1998; see also Chapter 26) and the USA (see

Chapter 22). Work is going on for Australia (see Chapter 25), China (a continuation of the work of Chen and Qiao 2000), Egypt (see el Mahdi 1999) and Amazonia (for Brazil see Machado and Fenzl 2000).

ATTRIBUTION TO SECTORS, ACTIVITIES AND FUNCTIONS

The throughput of the whole economy can be disaggregated and attributed to specific industrial sectors by 'top-down' approaches. This attribution[2] can be oriented towards economic or functional criteria. Usually on the basis of economic input–output (I/O) classification throughput of sectors may be determined by I/O analysis (see Chapter 10). This allows for an overall comparison of all industrial sectors. The sum of individual sectoral flows in general equals the economy-wide sum. Economic I/O tables are used to attribute physical inputs (Bringezu *et al.* 1998b) or outputs (Hohmeyer *et al.* 1997) of the national economy to the sectors of intermediate or final demand. Physical I/O tables (PIOT) provide a much more elaborate picture of sectoral product supply and delivery as well as resource inputs from the environment and waste disposal and emissions to the environment. PIOT have been established for Germany (Stahmer *et al.* 1998) and Denmark (Pedersen 1999).

The overall throughput may also be attributed to metabolic functions of the anthroposphere such as energy supply, nutrition, construction and maintenance (Bringezu 1997a). The attribution to 'activity fields' such as food supply, energy supply, construction, water supply and transport may be more actor-oriented but cannot simply be aggregated into one national account (Schandl and Hüttler 1997).

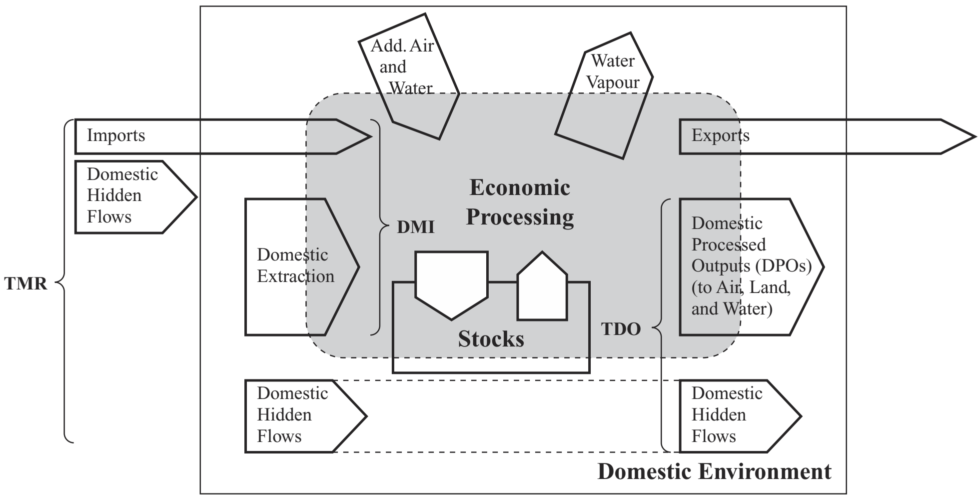
A 'bottom-up' approach may be applied to analyze the material flows of a specific sector. For instance, the flows of the construction sector had been approximated on the basis of various construction types. A comparison between 'bottom-up' and 'top-down' reveals significant differences (Friege 1997; Kohler *et al.* 1999). MFA of specific sectors often uses a combination of 'bottom-up' and 'top-down' methods and related data sources (for example, Glenck and Lahner 1997).

MFA-BASED INDICATORS

Material flow accounts provide an important basis for the derivation of environmental indicators and indicators for sustainability (Berkhout 1999; Jimenez-Beltran 1998; IME 1999). In order to monitor and assess the environmental performance of national and regional economies, a variety of indicator systems have been proposed (Moldan *et al.* 1997). The Driving Force-Pressure-State-Impact-Response (DPSIR) scheme was established as a framework (EEA 1999a, 1999b; OECD 1998a). (It had been used since the early 1990s as 'PSR' by the OECD.) The extraction of resources on the input side and the release of emissions and waste on the output side relate to environmental pressures, (sectoral) activities represent driving forces. The flows may change the state of environment which gives rise to various impacts and the societal or political response may influence the metabolic situation towards sustainability.

Corresponding to the different objectives in Table 8.1, indicators may focus on the specific impact per unit of flow (for example, emission of substances contributing to global warming) or on the volume of flows which exert a certain generic pressure (for example, consumption of water, energy, materials). MFA-based indicators have been introduced in official reports to provide an overview on the headline issues of resource use, waste disposal and emissions to air and water as well as eco-efficiency (EEA 2000; UKDETR 1999, Hoffrén 1999).

On the one hand, economy-wide material flow accounts provide a more comprehensive picture of the industrial metabolism than single indicators. On the other hand, they can be used to derive several parameters which – when taken in time series and for international comparison – provide certain aggregated information on the metabolic performance of national or regional economies (Figure 8.1). First international comparisons have been provided on input and resource efficiency indicators by Adriaanse *et al.* (1997) and on output and balance indicators by Matthews *et al.* (2000). (See also Chapters 15 to 17.)



Source: Matthews *et al.* (2000).

Figure 8.1 Economy-wide material flows

Input Indicators

Direct material input (DMI) measures the input of used materials into the economy, that is all materials which are of economic value and used in production and consumption activities; DMI equals domestic (used) extraction plus imports. Materials which are extracted by economic activities but that do not normally serve as input for production or consumption activities (mining overburden and so on) have been termed 'hidden flows' or 'ecological rucksacks'. Hidden flows (Adriaanse *et al.* 1997), or rucksack flows (Schmidt-Bleek *et al.* 1998; Bringezu *et al.* 1996) comprise the primary resource requirement not entering the product itself. Hidden flows of primary production are defined as unused domestic extraction or 'indirect material flows' (Eurostat 2000). Hidden flows of imports equal unused and used predominantly foreign extraction associated with the production and delivery of the imports. These are not used for further processing and are usually without economic value. DMI plus unused domestic extraction comprises total (domestic) material input.

Total material requirement (TMR)[3] includes, in addition to TMI, the upstream hidden material flows which are associated with imports and which predominantly burden the environment in other countries. It measures the total 'material base' of an economy, that is the total primary resource requirements of the production activities. Adding the upstream flows converts imports into their 'primary resource extraction equivalent'.

Data for TMR and DMI (including composition, that is input structure of the industrial metabolism) have been provided for China (Chen and Qiao 2000), Germany, the Netherlands, Japan, USA (Adriaanse *et al.* 1997), Poland (Mündl *et al.* 1999), Finland (Juutinen and Mäenpää 1999; Muukkonen 2000; FME 1999) and the European Union (Bringezu and Schütz 2001). DMI is available for Sweden (Isacsson *et al.* 2000). Work is going on for Italy (de Marco *et al.* 1999) and Amazonia (Machado and Fenzl 2000). TMI, although termed TMR, has been accounted for in Australia (Poldy and Foran 1999).

Output Indicators

Domestic processed output (DPO) represents the total mass of materials which have been used in the domestic economy before flowing into the environment. These flows occur at the processing, manufacturing, use and final disposal stages of the economic production–consumption chain. Exported materials are excluded because their wastes occur in other countries. Included in DPO are emissions to air from commercial energy combustion and other industrial processes, industrial and household wastes deposited in landfills, material loads in wastewater, materials dispersed into the environment as a result of product use (dissipative flows) and emissions from incineration plants. Material flows recycled in industry are not included in DPO.

Total domestic output (TDO) is the sum of DPO and disposal of unused domestic extraction. This indicator represents the total quantity of material outputs to the environment released in domestic territory by economic activity. *Direct material output* (DMO) is the sum of DPO and exports. This parameter represents the total quantity of direct material outputs leaving the economy after use, either into the environment or to the rest of the world. *Total material output* (TMO) also includes exports and therefore measures the total of material that leaves the economy; TMO equals TDO plus exports.

Consumption Indicators

Domestic material consumption (DMC) measures the total amount of material directly used in an economy, excluding hidden flows (for example, Isacsson *et al.* 2000). DMC equals DMI minus exports.

Total material consumption (TMC) measures the total primary material requirement associated with domestic consumption activities (Bringezu *et al.* 1994). TMC equals TMR minus exports and their hidden flows.

Balance Indicators

Net additions to stock (NAS) measures the physical growth rate of an economy. New materials are added to the economy's stock each year (gross additions) in buildings and other infrastructure, and materials incorporated into new durable goods such as cars, industrial machinery and household appliances, while old materials are removed from stock as buildings are demolished, and durable goods disposed of. NAS may be calculated indirectly as the balancing item between the annual flow of materials that enter the economy (DMI), plus air inputs (for example, for oxidization processes), minus DPO, minus water vapor, minus exports. NAS may also be calculated directly as gross additions to stock, minus the material outputs of decommissioned building materials (as construction and demolition wastes) and disposed durable goods, minus materials recycled.

Physical trade balance (PTB) measures the physical trade surplus or deficit of an economy. PTB equals imports minus exports. Physical trade balances may also be defined including hidden flows associated with imports and exports (for example, on the basis of TMC accounts).

Efficiency Indicators

Services provided or economic performance (in terms of value-added or GDP) may be related to either input or output indicators to provide efficiency measures. For instance, GDP per DMI indicates the direct materials productivity. GDP per TDO measures the economic performance in relation to material losses to the environment. Setting the value-added in relation to the most important inputs and outputs provides information on the eco-efficiency of an economy. The interpretation of these relative measures should always consider the trends of the absolute parameters. The latter are usually also provided on a per capita basis to support international comparisons.

Increasingly, MFA and its indicators will be used to provide the basis for political measures and to evaluate the effectiveness of such measures. For that purpose bulk material flow analyses and substance flow analyses can be combined and the monitoring of progress towards sustainability can be gradually improved by taking a stepwise approach (see Bringezu *et al.* 1998a).

NOTES

1. The indicative value depends on the relation to (a) other flows, (b) assessment parameters such as critical levels, and (c) system properties of the accounting (for example, systems borders from cradle to grave) (Bringezu 2000a).
2. To conform to LCA usage, attribution is sometimes called 'allocation'.
3 In studies prior to Adriaanse *et al.* (1997), TMR had been defined as total material input, TMI (for example, Bringezu 1997b).

9. Substance flow analysis methodology

Ester van der Voet

In this chapter, the analytic tool of substance flow analysis (SFA) is described and a general framework is presented to conduct SFA studies. SFA aims to provide relevant information for an overall management strategy with regard to one specific substance or a limited group of substances. In order to do this, a quantified relationship between the economy and the environment of a geographically demarcated system is established by quantifying the pathways of a substance or group of substances in, out and through that system. SFA can be placed in the scientific field of industrial ecology, as one way to operationalize the concept of industrial metabolism (Ayres 1989a). In this concept, an analogy is drawn between the economy and environment on a material level: the economy's 'metabolism', in terms of materials mobilization, use and excretion to create 'technomass', is compared to the use of materials in the biosphere to create biomass. The economy thus is viewed only in terms of its materials flows. The analysis of such flows in general is called MFA (material flow accounting or analysis). Udo de Haes *et al.* (1997) define SFA as a specific brand of MFA, dealing only with the analysis of flows of specific chemicals. MFA is a broader concept, also covering the analysis of mass flows through an economic system and the analysis of bulk flows of specific materials such as paper, glass and plastics and is treated in Chapter 8. The methodology is similar but the applications may be different. Mass and bulk flow studies provide macroeconomic indicators (von Weizsäcker *et al.* 1997; Adriaanse *et al.* 1997), while studies on flows of chemicals can be related to specific environmental problems and thus provide input for a pollutants policy.

The core principle of MFA and SFA is the mass balance principle, derived from Lavoisier's law of mass conservation (Lavoisier 1965 [1789]). This allows for various types of analysis, as will be elaborated below. Early applications can be found in ecology, such as for example by Lotka (1924) specifying nutrient budgets. A historic overview is given by Fischer-Kowalski (1998). The studies of biogeochemical cycles including the human interference, as practiced since the 1960s, could also be regarded as SFA (Miller 1968; Nriagu 1976; Lenikan and Fletcher 1977; Smil 1985; Schlesinger 1991). The study of flows in the economic system also starts in the late 1960s. Kneese *et al.* (1970) propose following flows and stocks of materials at a totaled level. Later examples are given by, among others, Baccini and Brunner (1991) and Adriaanse *et al.* (1997). The study of human-induced flows of chemicals are connected to the availability of specific resources (Meadows *et al.* 1972), but more specifically to pollution problems (Randers 1973; Kneese and Bower 1979; Ayres and Rod 1986; Stigliani and Salomons 1993). For roughly a decade, different efforts in the MFA/SFA field have been becoming more harmonized and there is a certain development towards a general methodology (Ayres and Simonis 1994; Adriaanse *et al.* 1997; Bringezu *et al.* 1997; Brunner *et al.* 1998; Hansen and Lassen 2000).

GENERAL FRAMEWORK

In general terms, materials flow studies comprise the following three-step procedure (van der Voet, Kleijn, van Oers, Heijungs, Huele and Mulder 1995): (a) definition of the system, (b) quantification of the overview of stocks and flows, and (c) interpretation of the results. All three steps involve a variety of choices and specifications, each of which depends on the specific goal of the study to be conducted, as will be argued below.

The first step in any materials flow study is to define the system. The system must be determined with regard to space, function, time and materials. If necessary, the system can be divided into subsystems. The various categories of processes, stocks and flows belonging to the system must be specified. Finally, this results in a flow chart: the specification of the *network of nodes*. To define the SFA system, a number of choices must be made with regard to the following aspects: spatial demarcation, functional demarcation, time horizon and materials to be studied.

The quantification of the network is the next step. This involves identifying and collecting the relevant data, on the one hand, and modeling, on the other. Three possible ways of modeling the system are briefly discussed in this chapter, all three types having their own data requirements, as well as their own potential for policy support: (a) accounting or bookkeeping, (b) static modeling and (c) dynamic modeling.

The quantified overview can be regarded as the result of an SFA study. However it is not easy to derive policy relevant conclusions from this. Two types of 'interpretation' are distinguished here: evaluation of the robustness of the overview quantification, and translating the overview into policy-relevant terms.

All of the above considerations are discussed in detail in the sections which follow.

SYSTEM DEFINITION

The general aim of most SFA studies is to provide the relevant information for a region's management strategy regarding specific chemicals. A number of choices regarding the system follow from this general aim.

Space and Function

In general, SFA studies will take a regional approach: specifying all flows of the selected materials within a certain geographical boundary, regardless of their function. Any size range from purely local (for example, 1 hectare) to global is possible. An administrative region such as a country or a group of countries has definite advantages, especially with regard to data availability (production and trade statistics). It also facilitates a linkage with a country's environmental policy. Other choices are possible, from the point of view of environmental analysis (river basins: for example, Ayres and Rod 1986; Stigliani and Anderberg 1994, or coastal seas), or otherwise (counties, polders, agricultural units, production plants). The inclusion of various scale levels within one study is also possible.

Time

Studies of flows automatically imply a time dimension, being expressed in mass units per time unit. In most SFA studies the time unit is one year. In some cases a shorter time period might be preferable, for example when variations within the year are also relevant, especially for environmental flows. In other cases a longer period could be more appropriate, for example when a slow stock-building process is being monitored.

Materials

A third choice involves the materials to be studied. Sometimes only one substance is studied at a time (for example, Anderberg, Bergbäck and Lohm 1989; Bergbäck, Anderberg and Lohm 1992; Annema *et al.* 1995; Ayres 1997b, 1998; Ayres and Ayres 1997, 2000; Brunner *et al.* 1998). Sometimes the object is a compound material or a coherent group of substances (Russell 1973; Russell and Vaughan 1976; Kleijn, Tukker and van der Voet 1997; Redle 1999). This choice, too, depends on the specific questions to be answered: materials studies may provide relevant information on the 'materials intensity' of a society, while one substance studies, on the other hand, are relevant for establishing the contribution of a society to specific pollution problems. Metals and nutrients are among the most frequently investigated substances.

For the remainder, the SFA system definition is derived from two basic notions regarding the relationship between the economy and the environment: the economy–environment distinction and the economy–environment integration.

The Economy–Environment Distinction

Differentiation into subsystems is not required for the modeling of substance flows, but it is useful for interpreting the results in terms of the economic–environmental interaction. Two subsystems are acknowledged: (a) the societal subsystem, also referred to as the economy, or (also used) the technosphere or the anthroposphere, which contains the stocks and flows mainly controlled or caused by humans; and (b) the environmental subsystem, also referred to as the biosphere. This subsystem contains the stocks and flows in the environment that can be described as biologically available.

The immobile stocks, encompassing the geological stocks in the geosphere or lithosphere, could be defined as a third, separate subsystem, with its own flows and processes (albeit on a geological time scale) and its own exchanges with the other two subsystems. Isolation from the economic or environmental surroundings is then the key criterion for characterizing stocks as immobile. Another possibility is to define the immobile stocks as a category of stocks within the other two subsystems. The geological stocks, as well as some environmentally harmless, inert bulk stocks such as atmospheric nitrogen gas and NaCl in the sea, or specific forms of long-term storage in the economy could then be encompassed. It is, of course, also possible to divide both subsystems into further subsystems. Husar (1994) for example distinguishes the biosphere, atmosphere and hydrosphere as subsystems of the environment. Within the economy, a distinction can be made between the various stages of a substance's life cycle, for example, or between the intentional and the unintentional flows of a substance. The usefulness of such subdivisions depends on the goal of the specific study.

The Economy–Environment Integration

The SFA method encompasses economic and environmental flows in a single system, so that the substance can be followed from its (economic) cradle to its (environmental) grave. In many cases, mostly owing to a lack of data, stocks are ignored and only flows (including accumulations, that is changes of stocks) are taken into account. Taking into account *all* flows and stocks, inspired by the regional approach, has the advantage that not only the intended effects, but also the unintended changes in substance flows resulting from certain societal developments, become apparent. A sectoral analysis is also possible.

The Substance Life Cycle

The system's flows are directed by economic and environmental processes. These processes are the main object of modeling. Each process, economic or environmental, is described in terms of transformations and flows. A great deal of modeling has been performed on the environmental processes: atmospheric dispersal, evaporation, leaching, runoff, bioaccumulation and so on. This has led to some very generalized models of the total environment, describing the exchange between the environmental compartments, such as the multimedia Mackay models (Mackay and Clark 1991). In fact, such a model can be incorporated in a general SFA model. A case in point can be found in Chapter 30 of the present volume (see also Guinée *et al.* 1999, on heavy metals, and Randers 1973, on DDT).

For the economic flows, Leontief-type input–output models (Leontief 1966) offer some possibilities (see, for example, Duchin 1998), despite drawbacks (notably aggregation in monetary terms). Nonetheless, the I–O systems approach can sometimes be used for SFA purposes. The economic processes together describe the substance's economic life cycle (Figure 9.1). Each of the life cycle stages is represented by a number of linked nodes (sectors or processes).

QUANTIFICATION OF THE OVERVIEW OF FLOWS AND STOCKS

The modeling of the system's substance flows and stocks can be attempted in different ways. Three are identified here:

1. accounting: keeping track of flows and stocks afterwards by registering them, thus enabling policy makers to spot trends, and to evaluate the effects of certain changes including policy measures;
2. static modeling: specifying the steady-state relations between stocks and flows (for example the Leontief I–O models).
3. dynamic modeling: including time as a modeling parameter, thereby making it possible to predict future situations.

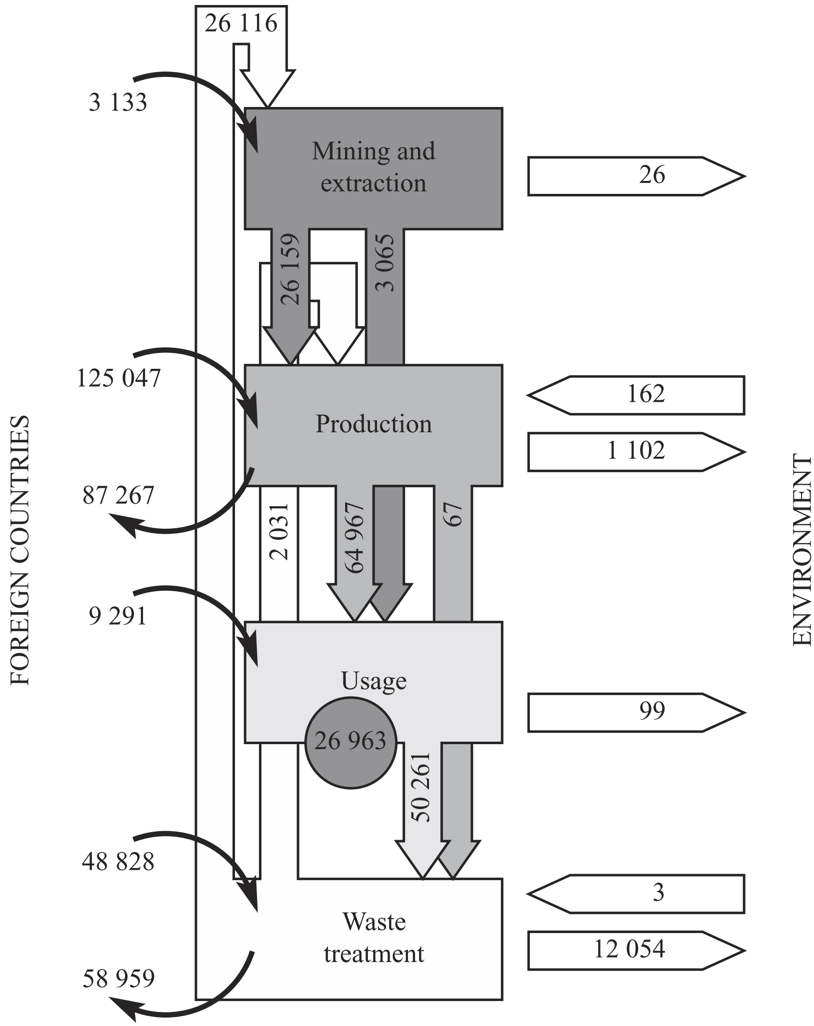


Figure 9.1 A substance life cycle for copper in the Netherlands, 1990 (kT Cu/year)

There is no 'best' choice: each type of modeling is useful, each has different functions for supporting environmental policy, and each has different data requirements.

Accounting

The first way to 'model' the system is to treat it as an accounting system. The input for such a system consists of data regarding the size of the system's flows and stocks of goods and materials, that can be obtained from trade and production statistics, and if necessary also data regarding the content of specific substances in those goods and materials. Emissions and environmental flux or concentration monitoring can be used for the environmental flows. A combination of these data together with application of the mass balancing principle then must lead to the desired overview of flows and stocks.

The accounting overview may also serve as an identification system for missing or inaccurate data. Missing amounts can be estimated by applying the mass balance principle. In this way, inflows and outflows are balanced for every node as well as for the system as a whole, unless accumulation within the system can be proved. This technique is most commonly used in materials flow studies, and can be viewed as a form of descriptive statistics (for example, Ayres *et al.* 1988; Olsthoorn 1993; Fleckseder 1992; Kleijn and van der Voet 1998; Palm and Östlund 1996; Tukker *et al.* 1996, 1997; Hansen and Lassen 2000). There are, however, some examples of case studies that specifically address societal stocks (Bergbäck and Lohm 1997; Bergbäck, Johansson and Molander 2000) and these are used as an indicator for possible environmental problems in the future.

Static Modeling

In the case of static modeling, the process network is translated into a set of linear equations describing the flows and accumulations as dependent on one another. Emission factors and distribution factors over the various outputs for the economic processes and partition coefficients for the environmental compartments can be used as such variables. A limited amount of accounting data is required as well for a solution of the set of equations, but the modeling outcome is determined largely by the distribution pattern. The description of the system as such a matrix equation opens up possibilities for various types of analysis: the existence of solutions, the solution space and the robustness of the solution can be studied by means of standard algebraic techniques, as is shown by Bauer *et al.* (1997) and by Heijungs (1994, 1997a) for the related product life cycle assessment.

Static Modeling and Origins Analysis

For a pollutants policy, insight into the origins of pollution problems is essential. The origins of one specific problematical flow can be traced at several levels (van der Voet *et al.* 1995c; Gleiß *et al.* 1998). Three levels may be distinguished:

1. direct causes, derived directly from the nodes balance (for example, one of the direct causes of the cadmium soil load is atmospheric deposition);
2. the economic sectors, or environmental policy target groups, directly responsible for the problem, identified by following the path back from node to node to the point of emission (for example, waste incineration is one of the economic sectors responsible for the cadmium soil load);
3. ultimate origins, found by following the path back to the system boundaries (for example, the import of zinc ore is one of the ultimate origins of the cadmium soil load).

With the origins analysis, a specific problematic flow can be traced back and it can be established which fraction comes from which process. The ultimate origins are the most difficult to trace, mainly because of the looped processes that occur within the system, and are the main concern of the modeled origins analysis. The origins analysis is made for a specific year. The set of equations, when solved, must therefore lead to an overview that is identical to the 'bookkeeping' overview for the same year. The account thus serves as a check, but could also be used to derive formulae. For the origins analysis, the set of equations is used in reverse: the specific problem flow is expressed in terms of the fixed variables. From this it follows that, for this purpose, all system inflows and nothing but the system inflows must be appointed as the fixed variables. All other flows are then dependent, either directly or indirectly, on the inflows. In some cases, this may run contrary to our perception of causality, but it is a necessary condition for a successful origins analysis.

Steady-state Modeling: the Effectiveness of Abatement Measures

The overview as obtained by accounting will rarely describe an equilibrium situation. In the economy as well as in the environment some stocks are building up while others decrease. Disequilibrium implies that the magnitude of flows and stocks, even with a constant management regime, is sure to change. Steady-state modeling aims at calculating the equilibrium situation belonging to a (hypothetical) substance management regime. It is not a prediction of a future situation. The result of steady-state modeling may instead be regarded as a caricature of the present management regime, not blurred by the buffering of stocks. It is therefore most suitable for comparisons between management regimes. In Chapter 30, a steady-state model is applied to the case of heavy metals to assess whether the present regime is sustainable. A comparison is made between the present management regime and a number of others, containing different sets of abatement measures. Static and steady-state models have been proposed by Anderberg *et al.* (1993) and applied for example by Schrøder (1995), Baccini and Bader (1996), Boelens and Olsthoorn (1998), van der Voet *et al.* (2000b) and also, for purely environmental flows, by Jager and Visser (1994).

Technically, steady-state modeling is roughly equivalent to comparative static modeling in economics, as well as to the type of modeling as used in environmental fate models (Mackay and Clark, 1991). The set of equations also contains distribution coefficients but the bookkeeping overview is not duplicated, since mass balance of flows is enforced at every node of the system and accumulation is ruled out. The outcome, within its limitations, is rather more robust than that of more sophisticated dynamic models, because many uncertainties are excluded.

Dynamic Modeling

For a dynamic model, additional information is needed with regard to the time dimension of the variables: the life span of applications in the economy, the half-life of compounds, the retention time in environmental compartments and so forth.

Calculations can be made not only on the 'intrinsic' effectiveness of packages of measures, but also on their anticipated effects in a specific year in the future, and on the time it takes for such measures to become effective. A dynamic model is therefore most suitable for scenario analysis, provided that the required data are available or can be estimated with adequate accuracy.

The main difference between static and dynamic SFA models lies in the inclusion of stocks in society (Ford 1999): substances accumulated in stocks of materials and products in households or in the built environment. Some studies are dedicated to the analysis of accumulated stocks of metals and other persistent toxics in the societal system (Gilbert and Feenstra 1992; Bergbäck and Lohm 1997; Baccini and Bader 1996; Fraanje and Verkuijlen 1996; also Lohm *et al.* 1997; Kleijn, Huele and van der Voet 2000). Such build-ups can serve as an 'early warning' signal for future emissions: one day, the stocks may become obsolete or recognizably dangerous (as has happened with asbestos, CFCs, PCBs and mercury in chlor-alkali cells). Then the stocks may be discarded and end up as waste and emissions. In some cases, this delay between inflow and outflow can be very long indeed. Bergbäck and Lohm (1997) also draw attention to stocks of products no longer in use, but not discarded yet: old radios or computers in basements or attics, out-of-use pipes still in the soil, old stocks of chemicals no longer produced, such as lead paint or pesticides and suchlike. They conclude that such 'hibernating stocks' could be very large. In order to estimate future emissions, which is a crucial issue if environmental policy makers are to anticipate problems and take timely action, it appears that such stocks cannot be ignored. There are basically two approaches to modeling the generation of emissions and waste flows from stocks. The first approach we denote as the *leaching* (or depreciation) model, the second as the *delay* model (Kleijn *et al.*, 2000). In the leaching model the generation of waste and emissions is represented as a fraction of the present stock. An emission or loss coefficient is defined and inserted in the substance flow model.

The delay model starts from the assumption that the output from societal stocks – the generation of waste and emissions – is determined by the past input into and by the residence time in the economy. The outflow in a certain year thus equals the inflow of a certain number of years earlier, this being the residence time. To build such a model, data are required on the present size as well as the historical build-up of societal stocks of substances, or – alternatively – on stock changes over past years. If available, such data can be used for building an empirical stock model. In practice such data bases are usually incomplete, and we must look for other ways to estimate stock behavior. One possible approach to estimating such stock behavior is to define stock characteristics (Ayres 1978; Van der Voet, Kleijn and Huppes 1995). Available data on the build-up and size of stocks can then be used to validate theoretical stock models and adjust their parameters if necessary. At present, this approach has been applied only as an example in a few specific cases (Olsthoorn *et al.* 1991; Kleijn *et al.*, 2000).

Both approaches are approximations, and the choice between them is likely to be dictated by data availability. Theoretical considerations may suggest that some stocks, or some emissions from stocks, should be treated differently from others. For example, leaching from landfills or corrosion of metal surfaces may be modeled most adequately by using simple coefficients, as in the leaching model, since the actual metal molecules leaching out first are not necessarily the ones first entering the stock. Once in the stock, each molecule has an equal chance of leaching out. For the discarding of products, on the other hand, it may be more appropriate to use the delay model, since products obviously have a residence time after which they enter the waste stage. In an integrated substance flow and stock model, both types of model may be required, depending on the type of outflow.

THE INTERPRETATION OF THE RESULTS FOR POLICY MAKERS

Substance flow analysis studies are designed to support environmental decision making. Although in practice such studies have been carried out successfully from the analysts' perspective, the implications for policy are not always clear-cut (Brunner *et al.* 1998). It would therefore seem appropriate to pay closer attention to the translation of SFA results into policy relevant terms. Three issues need to be addressed for such communication: the basic principles of SFA, the terminology and the complexity of SFA results. Regarding the *basic principles*, the usefulness of investigating societal metabolism is sometimes questioned. Policy makers often feel that knowledge of emissions and extractions is sufficient. Many publications have been devoted to the importance of studying societal metabolism since the publication of the concept of industrial metabolism by Ayres (1989a) and the political awareness of the role of societal flows and stocks as the instigators of environmental problems is slowly growing in the elaboration of policy principles such as 'integrated chain management' (Netherlands Ministry of Housing, Spatial Planning and the Environment 1991). *Terminology* is always a difficult issue in a relatively new area of investigation such as SFA. Even among scientists there is no established terminology in this area, which often leads to confusion. In addition, there is a lack of coherence between the scientific and the policy vocabulary. We can see, for example, that the policy concept of 'sustainability' has found its way into SFA research, but that it has become such a very broad concept that it covers virtually everything and has therefore been stripped of any precise meaning. In order to close the gap between policy and science, a more specific connection must be made: SFA scientists should point out more clearly the relevance of their results in terms of policy means and ends. The third obstacle is the *complexity* of the SFA results. These results are, in most cases, presented in an overview of flows and/or stocks connected with a given region. Often such an overview is too complicated to pinpoint precisely the relevant information. A further interpretation of the overview data is then required, also to avoid the risk of deriving spurious conclusions. It is tempting to streamline the results to make them more accessible by defining a *set of indicators*. By doing so different purposes can be served at the same time: reducing complexity as well as establishing a better connection with the language of policy.

Indicators play an important role in the interpretation of environmental data for environmental policy. The general idea is to aggregate from a large and ungainly dataset into a limited number of measures or yardsticks deemed to be relevant for environmental policy. Indicators are widely used by policy makers to measure developments in the state of the environment, human influence on the environment and the effectiveness of chosen policy measures. The concept 'indicator' is not strictly defined and in practice many widely differing things may serve as indicators. Several attempts have been made to establish a classification of indicators (for example, Opschoor and Reijnders 1991; OECD 1994a; Ayres *et al.* 1996; Azar, Holmberg and Lindgren 1996). It is also possible to define specialized or localized indicators as a part of a particular SFA study.

SFA indicators can be selected from the overview, by singling out a specific flow or stock as the relevant one to follow, or they can be calculated directly from the overview. Indicators may be defined for environmental flows and/or stocks, as an addition to the numerous environmental quality indicators already existing. Other possibilities are indicators for economic substance flows, or indicators for integrated chain management, which bear on (possible, future) losses from the economy to the environment; that is, 'leaks' out of the economic cycle. Examples include materials intensity, economic throughput, the technical or energy efficiency of groups of processes, secondary v. primary materials use and so on (Ayres 1997a). Another possibility is to compare economic mobilization of a certain substance with natural mobilization, as a measure of potential risk (Huele, Kleijn and van der Voet 1993). This goes in the direction of the study of biogeochemical cycles and their transformation by man's activity into anthropo-biogeochemical cycles.

Indicators should be designed to provide information of relevance for an integrated substance chain management policy, for example regarding (a) the existence and causes of environmental problems related to the substance; (b) the management of the substance chain or cycle in society; (c) early recognition of future problems and (d) the influence of policy measures, including both their effectiveness and various types of problem shifting. In addition, requirements can be defined for the indicators as a group, which must be suitable for evaluating an SFA overview for a specific year, but also for evaluating changes in flows and stocks over time as well as alterations thereof, as induced by environmental policy. Therefore a comparison between different regimes must also be possible. See also Guinée *et al.* (1999; also Chapter 30) and Moolenaar *et al.* (1997a, also Chapter 33) for agricultural soils and systems.

FINAL REMARKS

Substance flow analysis can be used in many applications. So far, there is hardly any standardization of the tool: systems definition, quantification and interpretation of the results can be, and in fact are, performed in many ways. Nevertheless, the steps to be taken and the choices to be made can be specified, as has been attempted in the above. A three-step framework has been defined: goal and systems definition, quantification of the overview of flows and stocks, and interpretation of the results.

In general, the outcome of the methodological choices will depend on the goal of the study and the questions to be answered. Whether or not to include environmental flows, in what detail to define the system's nodes, which sectors to include and whether or not to include stocks must be decided case-by-case. Something more can be said about the choice of the type of quantification, the second step of the general framework. Three options are presented: accounting, static modeling and dynamic modeling. It is important to realize that there is no 'best' choice: each type of modeling is useful and each has different functions for supporting environmental policy. The display below summarizes the possibilities for application of the three modeling types.

Type of quantification application	Accounting	Static modeling	Dynamic modeling
signaling	+		
spotting trends	+		
evaluation *ex post*	+		
origins analysis		+	
comparing regimes		+	
evaluation *ex ante*		+	+
extrapolating trends			+
scenario analysis			+

For the third step of the framework, the interpretation of the results, it is important to realize the limitations of the SFA tool. The conclusions to be drawn should fall into the boundary conditions of SFA and therefore necessarily be restricted to the life cycle of the investigated substance. Many other variables, such as impacts on the life cycle of other substances, costs or rebound effects, are also relevant. Some attempts have been made to combine different aspects within one model (for example, Kandelaars and van den Bergh 1997), some even successfully, but so far limited to very small systems. In the future it may become clearer whether this is a fruitful road to travel, or whether it makes more sense to leave the individual tools small and simple but use them together.

10. Physical input–output accounting

Gunter Strassert

A physical input–output table (PIOT) is a macroeconomic activity-based physical accounting system. A PIOT comprises not only the product flow of the traditional input–output table in physical units, but also material flows between the natural environment and the economy. Complete material balances can therefore be generated for the various economic activities (Stahmer, Kuhn and Braun 1997, p. 1).

Physical input–output accounting has many roots. Two main analytical strata can be distinguished; that is, production theory and national accounting. The former stratum is represented by Georgescu-Roegen (1971, ch. IX; 1979; 1984, p. 28) and Perrings (1987, pt I), both developing the physical economy–environment system, and the latter by Stahmer (1988, 1993), United Nations (1993a, 1993b), Radermacher and Stahmer (1998), Stahmer, Kuhn and Braun (1996, 1997, 1998). Both were interlinked and complemented by Daly (1968), Katterl and Kratena (1990[1]) and Strassert (1993, 1997, 2000a, 2000b, 2000d, 2001a). Physical input–output accounting was preceded by the concepts of materials/energy balance (Kneese, Ayres and d'Arge 1970; Ayres 1978, 1993a) and material flow accounting (MFA): see Chapter 14.

A first complete PIOT, that is, a macroeconomic material flow account in the form of an input–output table, was presented for Germany 1990 ('Old Länder') by the Federal Statistical Office (see Stahmer, Kuhn and Braun 1996, 1997, 1998). As statistical units of materials, tons are used. The original matrix comprises 58 production activities of the conventional monetary input–output accounting, plus an additional sector for external environmental protection services. In the meantime, the German input–output accounting has been revised repeatedly and the analytical concept has developed (see below). Another official national PIOT was established for Denmark in 1990 (Gravgård 1998). Other initiatives should be also be mentioned; for example, a derivative PIOT for a German Bundesland, Baden-Württemberg in 1990 (see Acosta 1998), a small national PIOT for Italy (Nebbia 1999) or an experimental three-sector PIOT for the USA in 1993. See Acosta (2000), who used revised flow charts for the major mass flows in the US economy, 1993, from Ayres and Ayres (1998).

CONCEPT OF A PHYSICAL INPUT–OUTPUT TABLE

A PIOT is a tabular scheme in which *n* activities ('production processes' or 'sectors') are represented by both their material inputs and outputs in physical units (for example, 1000 tons). The inputs are detailed by origin categories in the columns and the outputs are detailed by destination categories in the rows. Normally the same categories are used for both rows and columns, but it is possible to construct (non-square) matrices with different source and destination categories. Only the square matrix case is considered here. For ease of illustration, the input and output sides are considered separately. When the two parts (Tables 10.1a and 10.1b) overlap what results is the typical rectangular scheme of an input–output table with three quadrants (Table 10.1c). The fourth quadrant is omitted because it does not correspond to any 'real' economic transformation according to the intention of exclusive representation of activities with respect to the composition either of inputs or of outputs (formally speaking, it contains only so-called 'counter-bookings').

Table 10.1(a) Components of the input side of a PIOT

	P_1 … P_n			
Activities	• Secondary • (intermediate) • inputs	Primary inputs A	Primary inputs B	Total input
	P_n			

(b) Components of the output side of a PIOT

	P_1 … P_n			
Activities	• Secondary • (intermediate) • outputs	Final output A	Final output B	Total output
	P_n			

(c) Scheme of a PIOT with five components (I, IIA, IIB, IIIA, IIIB)

I Transformation matrix		II Final production A B
III Primary input A B		

Table 10.1 can be explained as follows. As compared with a traditional monetary input–output table (MIOT) in a PIOT the quadrants II and III are subdivided into two components, A and B, respectively. One may suppose that the components I, IIA and IIIA correspond to a MIOT (for modifications see below). Then, for a PIOT it is essential to add the material/energy components IIB and IIIB, which are omitted in a MIOT.

To be complete in terms of a material balance and to show the total production on the input side and the output side as well, in a PIOT it is necessary to include two components, primary input and final disposal. The direct inputs from nature may be in gaseous, solid or fluid form. These inputs are typical primary inputs because they are natural resources not produced within the economy (quadrant IIIB). The outputs of residuals, in terms of solid, fluid and gaseous residuals (quadrant IIB) are then added to the output side.

The extension of the primary input component to include the primary natural resources component (quadrant IIIB) is due to the fact that the products of the economic production system are only transformation products which require a corresponding provision of primary natural resources of low entropy. Without such a supply of energetic and material inputs the economic production system would not be viable, because it is not able to create these products itself. As these primary inputs cross the boundary of the economic production system, one can speak of quasi-imports.

In physical terms, economic production is defined as the transformation of a set of energetic and material inputs by a specific production activity into another set of energetic and material outputs. These outputs are either main products, included in the final output component (IIB), or joint by-products, so-called 'waste' (gaseous, solid, fluid residuals), included in the final production component IIB. As these final outputs cross the boundary of the economic production system back to the environment, one can speak of quasi-exports.

In a MIOT these inputs and outputs, although representing the greatest portion of total production, are excluded. From this it follows that a MIOT generally can only represent a relatively small part of total material production. Of course, it cannot meet the condition of a material balance.

A PIOT, as described above, represents the idea that in the economic production system, which is an open subsystem of a finite and non-growing ecosystem (environment), the economy lives by importing low-entropy matter energy (raw materials) and exporting high-entropy matter energy (waste) (Daly 1991a, p. xiii). Capital proper and labor are conceived of as funds or agents that transform the flow of natural resources into a flow of products. The added components on the input and on the output side represent the one-way flow beginning with resources and ending with waste, and can be thought of as the digestive tract of an open biosystem that connects them to their environment at both ends (Daly 1995, p. 151). In this sense, a PIOT is a descriptive scaffold for the one-way flow or 'entropic flow' through the economic production system (ibid.).

There are three other quadrants of a MIOT to consider: the transaction or transformation matrix (quadrant I), final production A, normally called final demand accounting (quadrant IIA) and primary input A, normally called value-added accounting (quadrant IIIA). In terms of the System of National Accounts (SNA) final demand corresponds to the gross national product account (consumption plus gross investment plus exports minus imports) and primary input A to the gross national income account including wages, interests, rents and profits and depreciation and public transfers.

In this physical context, instead of the monetary value-added accounting we have, as its physical counterpart, a physical fund-oriented accounting which includes the material input flows needed for maintaining the funds intact.[2] In the broader context of a PIOT, the services of the funds used as inputs for economic production can be recorded, even though they are not material. From this point of view, Stahmer introduced time units into physical input–output accounting (see Stahmer 1999).

A PIOT, because it incorporates materials balances, overcomes the conventional bias of national accounting which is based on the vision of the economic process as an isolated circular flow from firms to households and back again, with no inlets or outlets, (Daly 1995, p. 151). Hence the accounting is concentrated on the completeness of the materials balance and not on the correspondence of the final demand component (IIA) to the value-added component (IIIA) as in conventional (monetary) national accounting. Consequently, in a PIOT, households play a different role and can be included in the transaction matrix as a quasi-production activity.

EXAMPLE: A PIOT FOR GERMANY, 1990; A FUNCTIONAL SIX-SECTOR VERSION

An aggregated version of the original PIOT has been used to create Table 10.2. It has been modified to reflect aspects of a bioeconomic approach proposed by Georgescu-Roegen (1971, 1984) and modified by Strassert (1993, 1997). A general outline of this approach is shown here.

The transaction matrix includes the following six production activities:

M: procurement of raw materials for processing through extraction of matter in situ;
E: procurement of effective (available) energy (fuel) through extraction of energy in situ;
I: production of new capital goods (investment), capital fund (assets) and maintenance goods (servicing);
C: production of consumer goods for manufacturing and private households;
H: household consumption activities, transformation of consumer goods;
P: environmental protection services, collection and recycling of residuals in the same establishment and further treatment in external protection facilities or storage in controlled landfills.

For a first characterization of the physical production system of West Germany (‘Old Länder’, 1990) we look at the characteristic relations/shares which are represented in the total input column and total output row or, equivalently, in the corresponding (aggregated) production account (Table 10.3).

On the input side, starting from the bottom, we see that 78.8 per cent of the total consists of primary (raw) material inputs from nature; that is, natural resources in solid, fluid and gaseous condition, which are transformed by all production activities (including private households) into a set of outputs. Since the de-accumulations of stocks and fixed assets are only a tiny percentage, the total primary material input amounts to less than 79 per cent of total input. The remainder (about 21 per cent of total input) belongs to total secondary or transformation production; that is, the intermediate production of all production activities including that of private households.

What is the result? Starting now from the top on the output side, first comes intermediate production (about 21 per cent). Next comes final main products in terms of accumulation of stocks and fixed assets (material gross investment) plus exports minus imports, with a share of less than 1 per cent, and, third, total final by-production of material residuals or waste, in solid, fluid and gaseous condition, with a share of about 79.0 per cent.

Table 10.2 A physical input–output table for Germany, 1990 ('Old Länder'), six functional activities, million tons

		M	E	I	C	H
	Output	Extraction of matter in situ	Extraction of energy in situ	Production of capital goods	Production of consumer goods	Household consumption
	Input	1	2	3	4	5
1 M	Extraction of matter in situ	295.1	0.1	551.8	189.3	22.5
2 E	Extraction of energy in situ	3.8	38.1	0.6	292.9	2.5
3 I	Production of capital goods	0.2	0.2	8.3	7.7	4.6
4 C	Production of consumer goods	268.2	81.4	210.8	3 661.5	3 052.4
5 H	Household consumption	0.0	0.0	0.0	0.0	0.0
6 P	Environmental protection	20.4	0.0	21.1	35.7	0.2
7 II	Intermediate input	587.7	119.8	792.6	4 187.1	2 082.2
8 PI-A	Primary input A: de-accumulation	0.0	0.0	0.0	0.0	0.0
9 PI-B	Primary input B: materials from nature	1 705.4	1 985.2	389.3	41 627.2	280.4
	Gaseous	*640.0*	*1 151.5*	*168.9*	*0.7*	*0.0*
	Solid	*702.7*	*825.2*	*197.1*	*41 142.5*	*59.0*
	Fluid	*362.7*	*8.5*	*23.3*	*484.0*	*221.4*
10 TI	Total input	2 293.1	2 105.0	1 181.9	45 814.3	3 362.6

Table 10.2 (cont.)

		P	IO	FO-A	FO-B	TO	
	Output	Environmental protection	Intermediate output	Final output A: accumulation (investment) and net exports	Final output B: residuals (gaseous, solid, fluid)	Total output	
	Input	6	7	8	9	10	
1 M	Extraction of matter in situ	41.8	1100.6	10.3	1 182.2	2 293.1	1
2 E	Extraction of energy in situ	40.4	378.3	−115.7	1 842.4	2 105.0	2
3 I	Production of capital goods	214.2	235.2	603.7	343.0	1181.9	3
4 C	Production of consumer goods	1 579.5	8 853.8	10.1	36 950.4	45 814.3	4
5 H	Household consumption	2 647.6	2 647.6	12.6	702.4	3 362.6	5
6 P	Environmental protection	14.1	91.5	12.0	7 976.9	8 080.4	6
7 II	Intermediate input	4 537.6	13 307.0	533.6	48 997.3	62 837.3	7
8 PI-A	Primary input A: de-accumulation	19.9	19.9				
9 PI-B	Primary input B: materials from nature	3 522.9	49 510.1				
	Gaseous	*0.0*	*1 961.1*				
	Solid	*3 501.1*	*46 427.6*				
	Fluid	*21.8*	*1 121.7*				
10 TI	Total input	8 080.4	62 837.3				

Table 10.3 Production account of the German PIOT, 1990, million tons

Type	Input MT	Input %	Output MT	Output %
Intermediate production	13 307	21.2	13 307	21.2
Primary input A: de-accumulation (stocks, fixed assets)	20	0	533	0
Primary input B: (raw) material from nature	49 510	78.8	48 997	78.8
Solid	*1 961*	*3.1*	*1 648*	*3.1*
Fluid: process water	*6 041*	*9.6*	*6 125*	*9.6*
throughput water	*40 387*	*64.3*	*40 387*	*64.3*
Gaseous	*1 122*	*1.8*	*1 122*	*1.8*
Total	62 837	100	62 837	100

To get an overall picture, an efficiency indicator (e) can be used (for an ecological context, see Ulanowicz 1986; for an economic context, see Strassert 2000c). From the production account (Table 10.3) one can derive the gross production equation,

$$TPI + SI = SO + FPA + FPB \tag{10.1}$$

where

TPI = total primary input,
SI = secondary input,
SO = secondary output,
FPA = final production A,
FPB = final production B.

So

$$1 = FPA \,/\, TPI + FPB \,/\, TPI \tag{10.2}$$

or

$$1 = y + r \tag{10.3}$$

Efficiency (e) is defined as

$$e = 1 - r = y. \tag{10.4}$$

Using the numerical data from the production account (Table 10.3) efficiency (e) comes to:

$$e = 1 - 48.997/49.510 = 1 - 0.99 = 0.01 \tag{10.5}$$

The results presented support the hypothesis that the German economy can be characterized as a throughput economy (Strassert 2000a). The transformation capacity of the economy is still so low that the total primary input is almost totally transformed into residuals. This is true even if water is neglected.

With regard to national accounting, a complementary calculation is of interest. When we calculate the gross national production (GNP), according to the SNA definition as consumption plus investment plus exports, for the residuals we receive a share about 12 times higher than GNP. From this point of view, the focus is now on the transformation matrix, to find some characteristics of the pathways of the secondary (intermediate) production. In brief, because we are dealing with a highly linear order of production activities we have a straight pathway of material transformation where cycles are largely absent.

In general, cycles can be understood as a deviation from a strictly triangular input–output table (transformation matrix). In practice, the structure of input–output tables is a *mixtum compositum* ranging between two extremes, from the totally linear case on the one hand to the totally circular case on the other hand. It is assumed that a certain degree of linearity can be seen as a necessary working condition of a production system. A linear structure is inherent in almost every empirical input–output table and can be made visible (through 'triangularization'). Conversely, the same can be said of the degree of interdependence or circularity.

A triangular matrix is the result of the so-called 'triangularization'; that is, a systematic reordering of the j sectors such that out of a set of $p=j!$ (in our case $p=6!=720$) orders, in the matrix of the final order, the total of the values above the main diagonal is maximal. The triangularization method is generally applicable to quadratic matrices, say an input–output matrix or a voting matrix. This method has a long tradition in the context of economic input–output analysis. In a totally triangular matrix there are only zeros below the main diagonal, a situation which Roubens and Vincke (1985, p. 16) denote as 'total order structure'.

This case corresponds to a (strong) transitive overall final order of activities. Normally, the activities of a given input–output table are not ordered optimally for purposes of revealing the order structure. Thus triangularization can be understood as a method for testing and displaying the degree of achievement of a (strong) transitive overall order of activities.

After triangularization this degree, λ, called 'degree of linearity' in the context of input–output matrices, is defined as follows:

$$\lambda = \Sigma_{i<j}(C_{ij}) \,/\, \Sigma_{i\neq j}(C_{ij}) \qquad 0.5 \leq \lambda \leq 1 \tag{10.6}$$

The degree of interdependence is defined as

$$\delta = 2(1-\lambda). \tag{10.7}$$

As δ is the degree of 'feedback' or 'circularity', we have to take the complementary value $(1-\lambda)$. The factor 2 is chosen because the minimum value of λ is 0.5.

If we have only zeros below the main diagonal, then $\lambda=1$. In this case, the complementary 'degree of interdependence', δ, is minimized: $\delta=0$. The degree of linearity, λ, and the degree of interdependence, δ, combine as follows:

$$\lambda = 1 \text{ and } \delta = 0$$
$$\lambda = 0.5 \text{ and } \delta = 1$$

The German PIOT yields the following degrees:

degree of linearity: $\lambda = 0.96$,
degree of interdependence/circularity: $\delta = 0.08$.

These measures are near their extreme values (maximum/minimum); that is, the degree of linearity is very high and, conversely, the degree of interdependence/circularity is very low.[3] To present these results in a more meaningful form, the triangularized PIOT is filtered and transformed into Boolean form. Its elements are set equal to 1, if $x_{ij} > x_{ji}$, and equal to 0, otherwise. Table 10.4 shows the extremely linear organization of the production system; that is, when the activities are presented in the order E, C, M, I, H, P, the result is a complete triangular matrix. That means that the primary material input is transformed along this activity chain without any feedback circuits. Not even environmental protection services (activity 6) creates a feedback.

Table 10.4 Filtered triangularized PIOT

		E 2	C 4	M 1	I 3	H 5	P 6
2	E		1	1	1	1	1
4	C	0		1	1	1	1
1	M	0	0		1	1	1
3	I	0	0	0		1	1
5	H	0	0	0	0		1
6	P	0	0	0	0	0	0

This result, which is incompatible with the common idea of a recycling economy (at least in Germany), underlines the crude fact that the German economy is a typical throughput economy (see below).

CONCLUSION: DEFICIENCIES OF THE MATERIAL TRANSFORMATION SYSTEM

The conclusions presented in this section follow lines drawn by Ayres over a decade ago (Ayres 1989a). Compared with the production system of the biosphere, which has evolved to a nearly perfect system for recycling materials, our industrial production system has three main deficiencies.

Dependency on Non-renewable Resources

From the three salient characteristics, described by Ayres (1989a), that mark the difference between the naturally evolved biosphere and its human-designed industrial counterpart, the first is that the metabolic processes of biological organisms are derived (by photosynthesis) from a renewable source, sunlight (ibid., p. 34). In contrast, the energy input of our industrial system depends heavily on the extraction of non-renewable raw materials (fossil fuels). 'In this sense, the industrial system of today resembles the earliest stage of biological evolution, when the most primitive living organisms obtained their energy from a stock of organic molecules accumulated during prebiotic times' (ibid., p. 44). Here we have one of the origins of anthropogenic emissions.

Downstream Dominance

The second characteristic is that the metabolism of living organisms (cells) is executed by multi-step regenerative chemical reactions in an aqueous medium at ambient temperatures and pressures (Ayres 1989a, p. 39). In contrast to this capability, our industrial production system can be characterized as a throughput economy where the industrial processes are irreversible transformations. A low transformation intensity follows from this. It is low because the industrial processes differ from biological organisms in that they are not (yet) able to build complex molecules directly from elementary building blocks with relatively few intermediates, as, for example, by the citric acid cycle in each cell of a living organism (ibid., p. 43). Within multi-step regenerative chemical reactions, controlled by catalysts (enzymes), most process intermediates are regenerated internally within the cell (ibid., p. 39).

Generally, a cyclic organization of production processes is a necessary condition of self-reproducing systems. If, as in industrial production, such a cyclic organization is absent and the system is dominated by process chains, self-reproduction does not take place and intermediates are embodied in downstream products or immediately wasted.

Material Flush-out

'The third salient characteristic differentiating the biosphere from the industrial synthesphere is that, although individual organisms do generate process wastes – primarily oxygen in the case of plants and carbon dioxide and urea in the case of animals – the biosphere as a whole is extremely efficient at recycling the elements essential to life. Specialized organisms have evolved to capture nutrients in wastes (including dead organisms) and recycle them' (Ayres 1989a, p. 41).

These 'specialized decay organisms' (destruents) constitute a very important part of the biosphere, mainly of the pedosphere (soil), where they interact within a complex transformation network. For an analytical description see, for example, the input–output framework for the natural production system of Strassert (1993, 1997) where four main groups of soil organisms are represented as production/transformation activities which interlink two food chains, the saprophage and the biophage food chain, on two levels, the aerobic and the anaerobic soil level. This important transformation domain is the final complement of the cyclic organization of the material flow as a provision–transformation–restitution cycle (ibid.) which connects the last with the first domain; that is, the restitution domain with the provision domain.

The question arises as to whether, in our industrial production system, there exists such a transformation domain, the function of which can be compared with the final part of the digestive tract (Daly 1995, p. xiii) of the 'organisms biosphere'. The answer must be negative. Most of our industrial recycling activities are far too small in scale to achieve a comparable functional importance. Possibly the evolution of a new domain of industrial destruents in form of the specialized 'cracker' technologies is yet to come.

Open Conceptual Problems

In an early phase of physical input–output accounting it is quite natural that a lot of conceptual questions are still under discussion. An important example, out of a number of ambiguities, is the water problem. On the one hand, if the data are available, a comprehensive approach is preferred (the German case); that is, all water quantities are counted, including those directly related to a production process (process water) as well as those indirectly related (cooling or irrigation water). However, it can be argued that 'throughput' water has a minimal environmental impact (except where it is very scarce) and that data in many countries are poor. In this case a more restrictive approach, in principle oriented to process water, is suggested (Ayres 2000; Gråvgard 1998; Nebbia 1999).

The general problem an accounting scheme should (ideally) avoid is that the overall total of all materials is dominated by the quantities of water. This refers not only to raw materials and residuals, but also to products, which also include water sold to households by water supply enterprises. So when, as in the German case, roughly two-thirds of the total quantity of products of the economy in tons is domestic water and the overall content of water in gross output is about 92 per cent, a PIOT is in danger of presenting only a more or less impure water account.

From this point of view, several authors (Ayres *et al.* 2000; Gråvgard 1998; Nebbia 1999) propose a restrictive convention; namely that water participating in an economic process *only* as a passive carrier of heat or a diluent of waste should not be counted. On the other hand, water that participates actively in a chemical or biological process must be counted on both sides: that is, both as an input and as an output.[4]

Gråvgard proposes that the input of water be limited to the quantity of water supplied to (embodied in) products in the manufacturing industry, and which therefore leaves the industry again, together with the goods produced. Water supplied to products in agriculture, horticulture, forestry and fishery is implicitly included when calculating biomass weight. Additional water consumption, that is the water which evaporates on the output side or which the sectors discharge to the waste system and so on is not included (Gråvgard,1998, p. 9).

Nebbia (1999) too wants not to consider the water flow through the economic system, but only:

a. the amounts of water required, as 'process' water, during the production and transformation of goods (for example, required for the photosynthesis);
b. the amounts of water 'embodied' in the inputs;
c. the amount of water vapor released to air during the production and the use of commodities;
d. the amount of water used for drinking by animals and humans, as needed in the process of food metabolism (ibid., p. 5).

In contrast, the German approach is a comprehensive one. It is oriented to a complete picture of all material (mass) flows through the economic system, but in such a way that active and passive water are separated. In an actual and revised version of the German PIOT the primary input component comprises two corresponding water categories. Besides, a complementary own water account was presented from the beginning of physical input–output accounting.

Considering the different positions, the general problem arises, how to draw appropriate analytical borderlines of production processes and corresponding statistical units. In a sense, one can speak of a revival of an old debate in input–output theory concerning functional or institutional concepts of data representation.

From the point of view that 'every production system of any type whatsoever is a system of elementary processes' and that 'the concept of elementary process is well defined in every system of production' (Georgescu-Roegen 1971, p. 235), two different perspectives are possible; on the one hand, the perspective oriented to a selected elementary (say physical and chemical) process out of the set of elementary processes that constitute the overall production process of a firm or establishment, and the perspective oriented to the overall set of elementary processes of a firm or establishment, on the other hand.

Although both perspectives are related to a functional perspective, the latter perspective includes some organizational and institutional elements as is the case when an establishment is chosen as a basic statistical unit. This perspective, leaving aside practical statistical aspects and recording principles, has a proper justification insofar as, for example, all water is a complementary and therefore essential input, with the consequence that the transformation process cannot take place without it. This is independent of whether passive water, say cooling water, undergoes any transformation or not. In this context, one should remember that cooling water belongs to the material input flows needed for maintaining the funds intact.

Similar problems regarding conventions, albeit with different solutions, apply to air (excluding the air mass that 'accompanies' the flow of used oxygen, nitrogen and carbon dioxide: see Nebbia 1999, p. 6), overburden, crude metal ores and biomass in agriculture. (See, for example, Ayres and Ayres 1998.)

NOTES

1. A first attempt to establish a physical input–output table was made for Austria (Katterl and Kratena 1990) using input–output data for 1983. This pioneering study presented only incomplete results, especially with respect to primary inputs and final products.
2. A fund is defined as an agent in the sense of a natural or artificial system (worker, produced capital good, land) which is used and not consumed, as compared with a stock of goods which is accumulated and deaccumulated by flows. A flow is defined as a stock spread over time. A fund element enters and leaves the production process with its functional unit intact. A fund is a 'stock of services'. (For the production theoretical foundation of a 'flow-fund model', see Georgescu-Roegen 1971, ch. IX.)
3. As a complementary indicator the diagonal elements of the so-called 'Leontief inverse' $(\mathbf{I}-\mathbf{A})^{-1}$ can be used; insofar as the diagonal elements exceed unity the existence of circuits is indicated. In the German case all diagonal elements are very close to unity and therefore circuits are absent (for methodological explanations, see Strassert 2001b). Generally, it should be mentioned that the results also depend on the level of aggregation. In an early version of the German PIOT with nine activities there was also a comparable high degree of linearity, nevertheless some circuits could be identified (see Strassert 2000a, p. 325).
4. Therefore, the authors continue: 'This means that water and carbon dioxide consumed in photosynthesis, together with water vapor and carbon dioxide produced by respiration (as well as combustion) must both be included. The same is true of oxygen consumed by respiration and combination and generated by photosynthesis' (Ayres *et al.* 2000).

11. Process analysis approach to industrial ecology

Urmila Diwekar and Mitchell J. Small

Industrial ecology is the study of the flows of materials and energy in industrial and consumer activities, of the effects of these flows on the environment, and of the influence of economic, political and social factors on the use, transformation and disposition of resources (White 1994). Industrial ecology applies the principles of material and energy balance, traditionally used by scientists and engineers to analyze well-defined ecological systems or industrial unit operations, to more complex systems of natural and human interaction. These systems can involve activities and resource utilization over scales ranging from single industrial plants to entire sectors, regions or economies. In so doing, the laws of conservation must incorporate a number of interacting economic, social and environmental processes and parameters. Furthermore, new methods and data are required to identify the appropriate principles and laws of thermodynamics at these higher levels of aggregation (Ayres 1995a, 1995b).

Figure 11.1 presents a conceptual framework for industrial ecology applied at different scales of spatial and economic organization, evaluating alternative management options using different types of information, tools for analysis and criteria for performance evaluation. As one moves from the small scale of a single unit operation or industrial production plant to the larger scales of an integrated industrial park, community, firm or sector, the available management options expand from simple changes in process operation and inputs to more complex resource management strategies, including integrated waste recycling and re-use options. Special focus has been placed on implementing the latter via industrial symbiosis, for example, through the pioneering work of integrating several industrial and municipal facilities in Kalundborg, Denmark (Ehrenfeld and Gertler 1997). At higher levels of spatial and economic organization, for example, at national and, in recent years, global scales of management, policy may be implemented through the tools of regulation, economic incentives, taxation, trade policy and international agreements.

To evaluate the full range of options illustrated in Figure 11.1, highly quantitative information on chemical properties, thermodynamic constants and constraints are needed, as are data relating to firm, sector, national and global resource utilization and conversion. However, these data are often unavailable or difficult to obtain, and more qualitative, order-or-magnitude information must be developed and used. These different types of information are used in developing mass and energy balances, formulating process simulation tools and optimizing process designs. For the latter, multiple objectives and performance criteria must be considered. At the local scale, performance measures include conversion efficiency, throughput, cost and safety. While these factors remain applicable

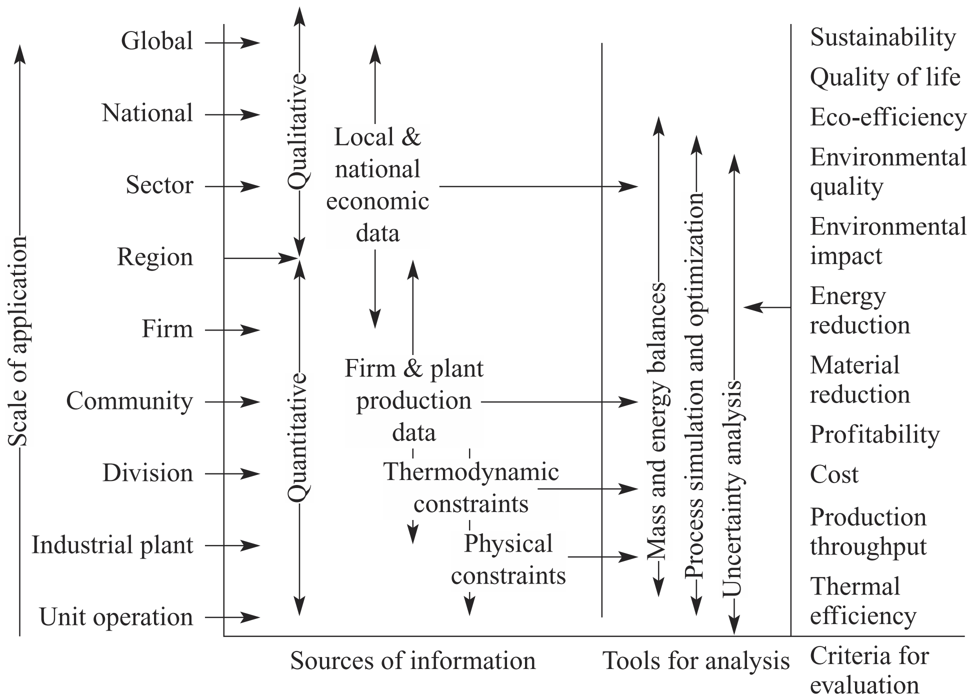


Figure 11.1 A conceptual framework for a process analysis approach to industrial ecology

at larger scales, broader metrics of overall resource use, the quality of the environment and the sustainability of economic activities are also considered. Industrial ecology provides a framework for integrating across these multiple scales of problem aggregation, assessment and analysis.

Process simulation provides a potentially useful basis upon which to begin to build assessments of this type. It is the purpose of this chapter to present an overview of the state of the art of methods for process simulation and optimization needed to develop a process-modeling approach to industrial ecology. To do this, we describe the development and state of the art of process simulators, their approach to mass and energy balance calculation, alternative methods for linking multiple unit operations and processes, and methods for estimating parameters, searching and optimizing the design space, and incorporating multiple, possibly conflicting, objectives and performance uncertainty.

Commercial process simulators were first made available in the late 1970s and have since been used extensively by the chemical process industry to track unit operation performance and component flows. Current simulators are equipped with detailed process and cost models, and include elaborate physical property databanks. However, they lack several capabilities needed to provide a complete economic and environmental assessment. Problem formulation and system representation for industrial ecology applications require the characterization of material, energy and information flows and reservoirs, often at a combination of local, regional and global scales. Even for a narrowly defined production process the necessary information for the full system may be highly dispersed among various organizations and organizational units (for example, see the analysis of a printed circuit board assembly process by Sheng and Worhach 1998). Such problems are only multiplied when dealing with multiple firms, industrial sectors or whole economies, and multiplied again when environmental impacts are added to the equation. A multi-objective approach to design under uncertainty is proposed to begin to address these assessment challenges.

PROCESS SIMULATION: AN ECOLOGICAL PERSPECTIVE

Process simulation involves the utilization of computer software resources to develop an accurate, representative model of a chemical process in order to understand its behavior in response to different inputs and control. In the past, process simulation was mainly concerned with the development of sophisticated unit operation blocks to predict mass flows of principal components through a process. In recent years, environmental consciousness has led to demands for tracking trace components (for example, resulting from fugitive emissions) that have an impact on environmental health and compliance, as well as major product and process components. Coupled with this demand for higher resolution models is the need for sophisticated computer-aided process design tools to identify low-cost, environmentally friendly solutions in the presence of considerable uncertainty. This calls for an integrated hierarchy of models, including modules with a high degree of detail for individual unit operations and process engineering activities, to simpler modules for analyzing system interactions at higher scales, with material flows and symbiotic interactions often controlled by exogenous factors, market forces or government regulation.

Many industries, both private and public, are involved in the transformation of raw material to useful products and by-products (some of which may be environmentally unacceptable). Several use process simulation tools to model their core production processes. These include chemical industries involved in the processing of organic and inorganic materials, the electric power industry involved in the transformation of fossil fuel to energy, and municipal treatment plants involved in the transformation of dirty to clean water. Effective facility operation is dependent upon accurate process simulation for assessing the material and energy flows through the process, so that the required thermal, environmental and economic performance can be assessed. These same process simulation tools have the potential to address programs and strategies to improve material and energy flows at higher scales of economic aggregation, providing guidance for industry, governments and citizens wishing to improve efficiency, sustainability and environmental quality through pollution prevention, material re-use, waste recycling, and material and energy conservation.

Process Simulation Tools

To understand how process simulation is used to model and design complex systems, the key components of a process simulation software package are identified and reviewed. The essential building blocks of a process simulator or 'flowsheeting' package include the following:

- Thermodynamic models: these are models developed to predict the different physical properties of the components under process conditions.
- Unit module models: these are routines that simulate the different unit operations (distillation, mixing, splitting, heat exchange and so on) and processes (reactions, mass and energy transfer, head loss).
- Data bank: the data on component physical properties, reaction rates and cost coefficients.

In addition to these, there are mathematical routines for numerical computations and cost routines for performing an economic analysis of the process.

Process simulation software can be classified as 'sequential modular', 'equation-oriented' or 'simultaneous modular'. Traditionally, most simulators have adopted a sequential modular approach. With this approach, individual modules are developed for each unit operation and process. Output stream values are computed for each, given the input stream values and the equipment parameters. Each unit module in a flowsheet is solved sequentially. The overall flowsheet calculations in a sequential modular simulator follow a hierarchy. Thermodynamic models and routines are at the bottom of this hierarchy, followed by the unit operation modules that perform the necessary material and energy balances, based on the thermodynamic property routines. At the next level design specifications dictate iterative calculations around the units, superseded by the recycle iterations for stream convergence. Program utilities, such as parameter estimation and optimization, occupy the highest level in the calculation hierarchy in the sequential modular framework.

Equation-oriented simulators define and solve a set of simultaneous non-linear equations that represent the process modules, mass and energy balances in the process. Although these simulators are more flexible in terms of information flow, they are more difficult to construct, and it is often difficult to diagnose errors when they occur. The simultaneous modular approach utilizes individual modules for each unit operation and process, as in the sequential modular approach, but attempts to establish a more immediate link among the inputs, outputs and operations of these individual modules. This is accomplished by defining a set of linear equations that approximately relate the outputs for each module to a linear combination of its input values. These equations are solved simultaneously in the simultaneous modular approach.

While efforts are under way to develop and advance equation-oriented and simultaneous modular software systems for education and research applications, most of the currently available, widely applied commercial simulators are sequential modular in nature. However, as indicated in Table 11.1, a significant effort has been made in recent years to develop and disseminate equation-oriented packages. There are no commercial simulators that use the simultaneous modular approach as yet.

Process simulators are also classified on the basis of their level of temporal aggregation; that is, whether the processes being considered are steady-state or dynamic in nature. Accordingly, steady-state and dynamic simulators are both available for modeling continuous processes. The sequential modular simulators shown in Table 11.1 are steady-state simulators. The equation-oriented simulators in the table can be used for both dynamic and steady-state analysis, but are mostly used for dynamic simulations.

The following example illustrates the use of ASPEN, a sequential modular simulator, to model the steady-state behavior of a benzene production process.

Table 11.1 Process simulation tools

Simulation package	Type
FLOWTRAN	Sequential modular
FLOWPACK II	Sequential modular
PRO II (previously PROII)	Sequential modular
ASPEN Plus	Sequential modular
SPEEDUP	Equation-oriented
ASCEND	Equation-oriented
MODELLA	Equation-oriented
gPROMS	Equation-oriented

Modeling Benzene Production

The hydrodealkylation (HDA) of toluene to produce benzene is often used as a benchmark for demonstrating chemical process synthesis methods. The HDA process has been extensively studied by Douglas (1988) using a hierarchical design/synthesis approach. The problem presented and solved here is based on the flowsheet structure analyzed by Diwekar *et al.* (1992), which involved the selection of the flowsheet configuration and some of the operating conditions that maximize profit. The flowsheet for this case study is described below.

The primary reaction of the HDA process is

$$C_6H_5CH_3 + H_2 \rightarrow C_6H_6 + CH_4.$$

In addition to this desired reaction, an undesired reaction

$$2C_6H_6 \leftrightarrow C_6H_5 + H_2$$

also occurs. These homogeneous gas phase reactions occur in the range of 894 °K and 974 °K. A molar ratio of at least 5:1 hydrogen to aromatics is maintained to prevent coking. The reactor effluents must be quenched to 894 °K to prevent coking in the heat exchanger following the reactor.

The HDA flowsheet is shown in Figure 11.2. In this process, benzene is formed by the reaction of toluene with hydrogen. The hydrogen feed stream has a purity of 95 per cent (the rest is methane) and is mixed with a fresh inlet stream of toluene, a recycled toluene stream and a recycled hydrogen stream. The feed mixture is heated in a furnace before being fed to an adiabatic reactor. The reactor effluent contains unreacted hydrogen and toluene, benzene (the desired product), diphenyl and methane; it is quenched and subsequently cooled in a flash separator to condense the aromatics from the non-condensable hydrogen and methane. The vapor stream from the flash unit contains hydrogen that is recycled. The liquid stream contains traces of hydrogen and methane that are separated from the aromatics in a secondary flash unit. The liquid stream from the secondary flash unit consists of benzene, diphenyl and toluene. It is separated in two distillation columns. The first column separates the product,

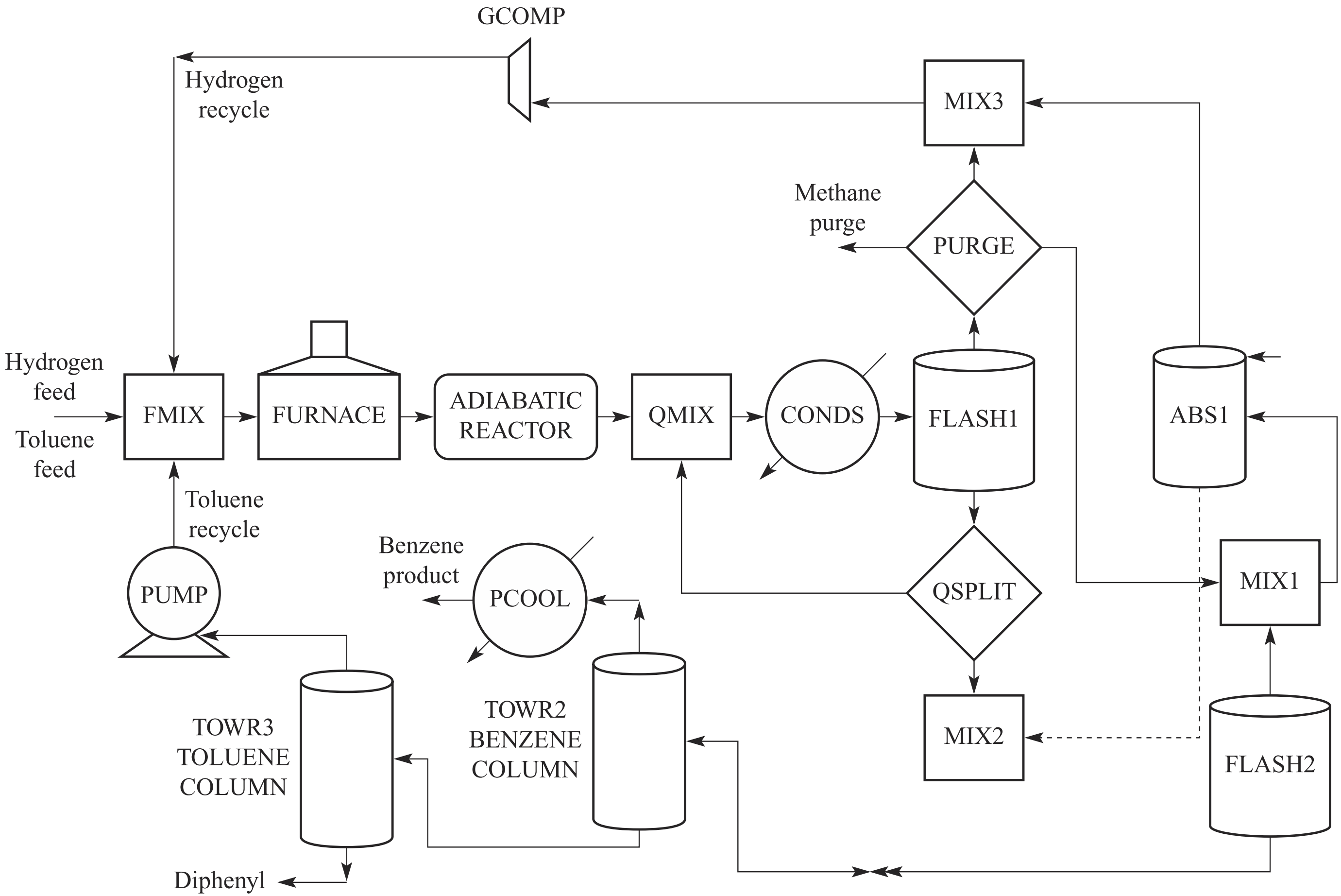


Figure 11.2 The process flowsheet for the production of benzene through the hydrodealkylation of toluene

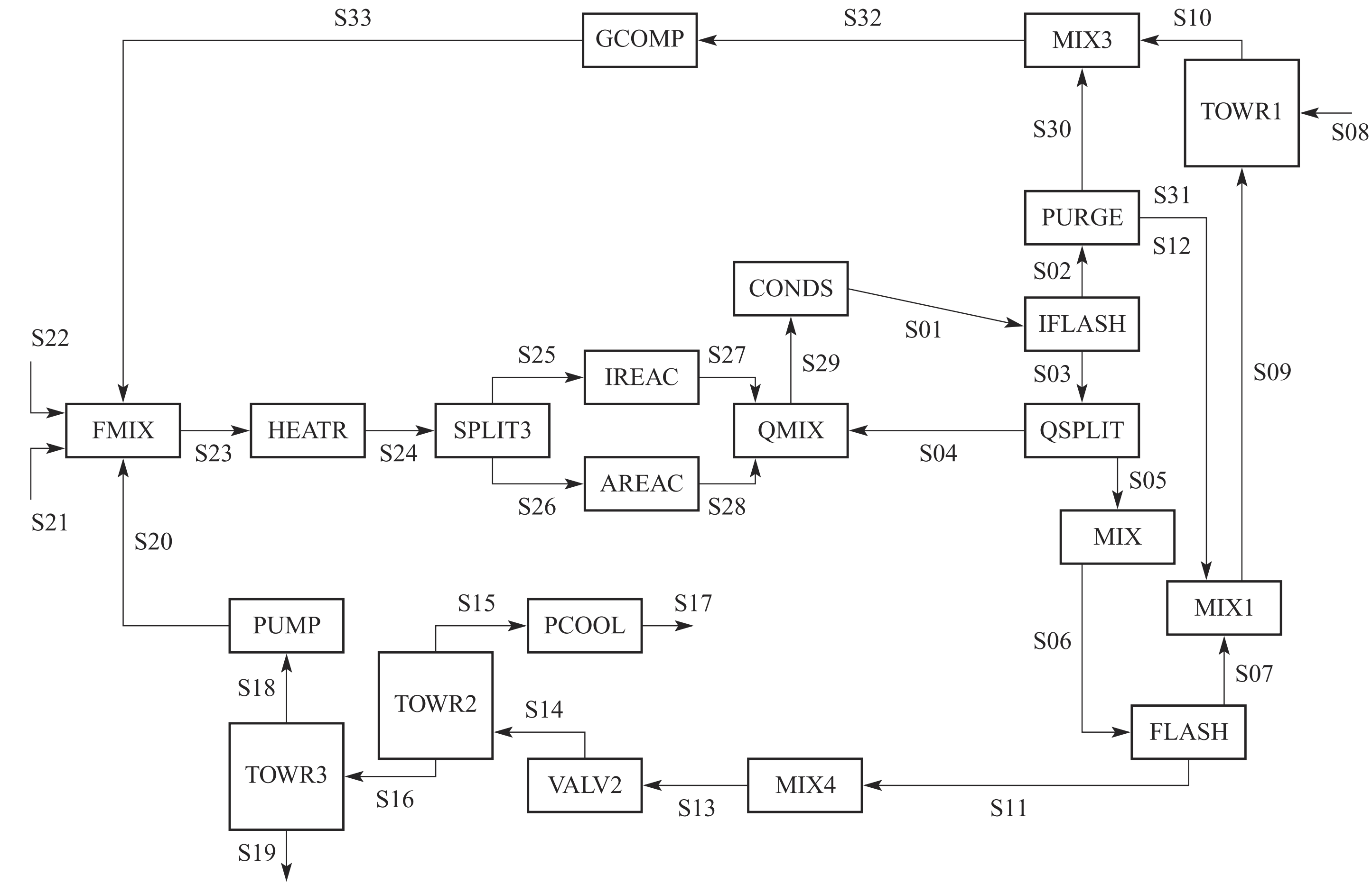


Figure 11.3   ASPEN representation of the HDA process

benzene, from diphenyl and toluene, while the second separates the diphenyl from toluene. The toluene is recycled back into the reactor.

Figure 11.3 presents the ASPEN representation of this flowsheet where unit operation blocks, including splitters, separators and reactors, are used as building blocks to track the material and energy streams through the complete process. Material and energy balances are computed around each unit and the system state variables are calculated, including component flows and system thermodynamic properties like enthalpy, entropy and so on, as shown in Table 11.2.

Table 11.2 Sample results for the HDA flowsheet simulation

Unit Operation Block Results

FLASH:2-OUTL (FLASH2): FLASH
INPUT STREAM(S): S01
OUTPUT STREAM(S): S02 S03
PROPERTY OPTION SET SYSOP0

Conventional components		Mass and Energy Balance In	Out	Relative diff.
H_2	LBMOL/HR	2047.48	2047.41	3.66387e-05
CH_4	LBMOL/HR	2414.51	2414.54	−1.39574e-05
C_6H_6	LBMOL/HR	374.131	374.139	−2.16750e-05
C_7H_8	LBMOL/HR	227.842	227.837	2.21647e-05
$C_{12}H_{10}$	LBMOL/HR	16.8401	16.8394	4.22304e-05
Total balance				
MOLE	LBMOL/HR	5080.8	5080.76	7.66967e-06
MASS	LB/HR	95679.1	95679.5	−4.68125e-06
ENTHALPY	BTU/HR	−6.60813e+07	−6.60813e+07	1.48284e-05

Stream results

Stream ID		S01	S02	S03
From:		CONDS	FLASH	FLASH
To:		FLASH	PURGE	QSPLIT
H_2	LBMOLE/HR	2047.4828	2046.7644	0.6433
CH_4	LBMOLE/HR	2414.5081	2390.8028	23.7389
C_6H_6	LBMOLE/HR	374.1306	17.8004	356.3382
C_7H_8	LBMOLE/HR	227.8422	3.5885	224.2486
$C_{12}H_{10}$	LBMOLE/HR	16.84	0.096816	16.8336
TOTAL	LBMOLE/HR	5080.8038	4458.9569	621.8079
TEMP	DEGREES F	100	100	100
PRES	PSIA	465	465	465
ENTHALPY	BTU/LBMOLE	−13006	−16927	15105
V	FRACTION	0.8776	1.0	0.0
L	FRACTION	0.1223	0.0	1.0
ENTROPY	BTU/LBMOLE-R	−21.6285	−15.6219	−64.704
DENSITY	LBMOLE/CUFT	0.0463	0.0774	0.6473
AVG MW		18.8314	9.9133	82.785

Consider the extension of this simulation technology to track flows in a multi-plant or multi-sector analysis to achieve industrial symbiosis. Now, instead of unit level balances, each plant or sector is represented as a building block for the complete analysis. This can be achieved either by simplifying each process as a simple reactor unit operation block in a process network, as done by Chang and Allen (1997) in their analysis of 428 chemical processes that produce or consume 224 chemicals, or by using detailed recipes from chemical engineering textbooks and carrying out detailed elemental balances (Ayres and Ayres 1997). While the first approach offers a simplified solution to a large, complex problem, the second approach can address the problem of data inconsistencies and dispersion by constraining the system. This approach is described next.

THERMODYNAMIC AND OTHER CONSTRAINTS

One of the major problems in including industrial ecological concepts in design is the problem of data inconsistencies and dispersion. Even for a narrowly defined production process, the necessary information is highly dispersed and in various forms. These inconsistencies can be attributed to one or more of the following factors: (a) non-comparable units of measurements; (b) uncertainties in the assumptions; (c) confidential and non-verifiable data and data from unreliable sources; (d) measurement uncertainties and (e) data violating laws of physics.

The first law of thermodynamics for conservation of mass and energy is applicable to every process network. It is, therefore, applicable to every firm and every industry that is in a steady state. This means that, for every process or process chain, the mass inputs must equal the mass of outputs, including wastes. Moreover, in many processes, non-reactive chemical components, such as process water and atmospheric nitrogen, can also be independently balanced. Thus various independent material balance constraints may have to be satisfied for each process. In short, systematic use of material balance conditions can increase the accuracy of empirical data by reducing error bounds (Ayres 1995a, 1995b). Alternatively, the material balance conditions can be used to 'fill in' missing data. Furthermore, material balance conditions are not the only basis for data augmentation. Energy conservation, constitutive relationships or statistical methods can also be used.

Process simulators are based on mass and energy balance principles. They utilize thermodynamic models and data, and hence are ideally suited for imposing these constraints on the available data. However, the constraints and data involved are not restricted to mass and energy balance principles, and are available in various forms. For example, it is common practice to report undetectable quantities of emissions in terms of the detection limit (or least count) of the measuring instrument (specifying that the data may be less than or equal to the detection limit). Sometimes the data are reported in order-of-magnitude terms (for example, refer to Case 3 in Ayres 1995a, where the Benzo(a)pyrene content is reported to be much smaller than 0.0001). Furthermore, discrete, categorical information about the occurrence or non-occurrence of particular reactions, or the presence or absence of reaction by-products, may be available.

Given that knowledge is available in various forms (for example, quantitative models for material and energy balances, order-of-magnitude information, qualitative information and logical information), a unified framework that incorporates information of each type in its inference is desirable. Optimization methods combined with artificial intelligence techniques, as proposed in Kalagnanam and Diwekar (1994), provide such a framework, in which information can be represented as inequality constraints. Unlike numerical methods for solving equations (equality constraints), optimization methods can handle both equality and inequality conditions and hence can be used to make inferences from data in various forms.

Defining Objectives and Goals

As stated earlier, methods for assessing economic impacts and profitability have been available for a number of years. However, methods and measures for characterizing environmental impacts and sustainability are as yet in their infancy. Recent attempts at defining ecological impacts for use in life cycle assessment and similar industrial ecology applications include the environmental burden system by ICI (Wright *et al.* 1998), sustainability indicators by Tyteca (1999), ecological risk indicators described by Koenig and Cantlon (2000), exergy as a unifying indicator for material and energy transformation (Ayres 1995b), environmental damage indices (DeCicco and Thomas 1999) and the generalized waste reduction (WAR) algorithm (Cabezas *et al.* 1997; Cabezas *et al.* 1999; Young and Cabezas 1999). The WAR algorithm uses a series of indices characterizing different environmental, social and economic impacts. With WAR the potential environmental impact is defined in terms of the pollution index, calculated by multiplying the mass of each pollutant emitted by a measure of its potential impact, then summing over all pollutants. This index is a carefully constructed function encompassing a comprehensive list of human health and environmental impacts for each chemical (see Table 11.3). However, like the other methods described above, the WAR index provides a highly simplified representation of environmental impacts. For example, effects of pollutants emitted to different media are not differentiated in the WAR algorithm. Chemical exergy content likewise provides only a partial insight into environmental impact, since it cannot be directly linked to toxicity to humans or other organisms. Nonetheless, these impact assessment methods provide a first-order qualitative indication of the environmental damage and hence a useful starting point for analysis.

Table 11.3 The potential environmental impact categories used within the WAR algorithm

Local Toxicological		Global Atmospheric	Regional Atmospheric
Human	Ecological		
Human toxicity potential by ingestion (HTPI)	Aquatic toxicity potential (ATP)	Global warming potential (GWP)	Acidification, or acid rain potential (ARP)
Human toxicity potential by exposure, dermal and inhalation (HTPE)	Terrestrial toxicity potential (TTP)	Ozone depletion potential (ODP)	Photochemical oxidation potential or smog formation potential (PCOP)

Recently the WAR algorithm was added to the ASPEN simulator to allow consideration of the eight environmental impacts shown in Table 11.3. This was easily done, since chemical simulators keep track of mass balance and emissions information required for calculation of these indices. Similarly, the unified indicator based on exergy proposed by Ayres (1995b) is readily computed using process simulation technology, since most commercial simulators have a unit operation block based on the 'concept of Gibbs free energy minimization'.

Once different environmental impacts are calculated, they must be weighted and balanced against each other, as well as other concerns, such as cost and long-term sustainability. These multiple, often conflicting, goals pose significant challenges to process optimization and design. How can designs be identified that best satisfy multiple objectives? Multi-objective optimization algorithms provide a particularly useful approach, aimed at determining the *set* of non-dominant/non-dominated ('Pareto') designs where a further improvement for one objective can only be made at the expense of another. This determines the set of potentially 'best' designs and explicitly identifies the trade-offs between them. This is in contrast to cost–benefit analysis, which deals with multiple objectives by identifying a single fundamental objective and then converting all the other objectives into this single currency. The multi-objective approach is particularly valuable in situations where there are a large number of desirable and important production, safety and environmental objectives which are not easily translated into dollars. Formulation of a process simulation and optimization model with multiple objectives is illustrated in the following section, with particular application to the HDA benzene synthesis problem.

ECOLOGICAL AND ECONOMIC CONSIDERATIONS: A MULTI-OBJECTIVE OPTIMIZATION PROBLEM

As stated earlier, algorithms such as WAR provide a first approximation of environmental objectives. However, the various environmental impact indices and economic objectives are so different in terms of evaluation, quantification and magnitude that the choice of relative weights for environmental and economic impacts depends upon the decision makers' perspectives. Thus it is necessary to provide decision makers with the complete economic and environmental surface, so that they can understand the full set of alternatives and the trade-offs among them in terms of the desired objectives.

A Multi-objective Optimization Framework

As is well known, mathematics cannot isolate a unique optimum when there are multiple competing objectives. Mathematics can at most aid designers to eliminate alternatives dominated by others, leaving a number of alternatives in what is called the 'Pareto set' (Hwang *et al.* 1980). For each of these alternatives, it is impossible to improve one objective without sacrificing the value of another, relative to some other alternatives in the set. From among the dominating solutions, it is then a value judgment by the customer to select which alternative is the most appropriate. At issue is an effective means of finding the members of the Pareto set for a problem, especially when there are more than two or three objectives, the analysis per design requires significant computations to complete and there are a very large number of feasible alternatives.

For example, consider the generalized WAR algorithm which expresses environmental objectives in terms of potential impact indices combined together using weighting factors, a_i. This formulation used in the WAR algorithm can be easily expressed in terms of the weighting method for a multi-objective optimization problem where different weights are assigned to obtain the Pareto surface. The generalized formulation is shown below:

$$\text{Min } \dot{I}^{(NP)}_{out} = \sum_{i=1}^{EnvCat} \alpha_i \dot{I}_i^{(NP)}$$

subject to:

mass and energy balance constraints
decision variables bounds

where $\dot{I}_i^{(NP)}$ is the rate of potential environmental impact generation for all the *NP* products and $\dot{I}^{(NP)}_{tot}$ is the weighted sum across these.

The basic strategy of the weighting method is to transform the multi-objective problem into a series of single objective problems with weighting factors assigned to each objective. The Pareto set can be derived by solving the large number of optimization problems created by modifying the weighting factors of the objectives. However, the major disadvantage of using the weighting method is its inefficiency that arises because of the large number of optimization problems that must be solved for the different linear combinations of objectives. It is also difficult to direct and limit the search to the region of the non-dominated surface which the decision maker prefers.

The constraint method is another technique for generating an approximation of the Pareto set. The basic strategy also is to transform the multi-objective problem into a series of single objective problems. A purely algorithmic solution approach (Cohon 1978) is to select one of the objectives to maximize (for example, profit) while each of the other objectives (for example, potential environmental impacts) is turned into an inequality constraint with a parametric right-hand side ($\varepsilon_1, \ldots, \varepsilon_p$). It is important to note that each optimal design in the Pareto set derived from a combination of α_i ($i = 1, \ldots, EnvCat$) by the weighting method, can be alternatively generated from a corresponding combination of ε_i ($i = 1, \ldots, EnvCat$) by the constraint method. One is a mapping of the other. For example, the upper bound of ε_i used in the constraint method correspond to $\alpha_i = 0$ in the weighting method, and the lower bound of ε_i used in the constraint method corresponds to $\alpha_i = +\infty$ in the weighting method. In other words, the upper and lower bounds of ε_i cover the entire range between 0 and $+\infty$ for α_i. The constraint method offers the advantages of better control over exploration of the non-dominated set and of locating points anywhere along the non-dominated surface.

The constraint method based on profit

$$\textbf{max} \qquad \textit{Profit}$$

subject to:

$\dot{I}_i^{(NP)} \leq \epsilon_i, i = 1, \ldots, EnvCat$
mass and energy balance constraints
decision variables bounds

By selectively decreasing each ε_i and rerunning the optimization (resulting in a lower maximum profit), the analyst explicitly identifies the trade-off between the profit that must be forgone to achieve improved environmental performance in environmental category i (that is, lower $\dot{I}_i^{(NP)}$). Of course, such trade-offs only occur on this outer envelope of the Pareto surface; the method generates designs where *mutual* improvement in both environmental quality and profit have been achieved to the maximum extent possible.

Determination of the Pareto set for various potential environmental impacts and economic objectives, using either the weighting or the constraint method, requires solution of a large number of optimization problems. For example, if there are six objectives and five of them are evaluated over 10 levels, we must solve 100 000 optimization problems. To circumvent this problem, a new and efficient multi-objective optimization algorithm based on the constraint method has been developed (Fu and Diwekar, 2001). Figure 11.4 illustrates the major features of this algorithm. In the outer loop, problem inputs are specified and the optimization problem is defined in terms of the objective function and

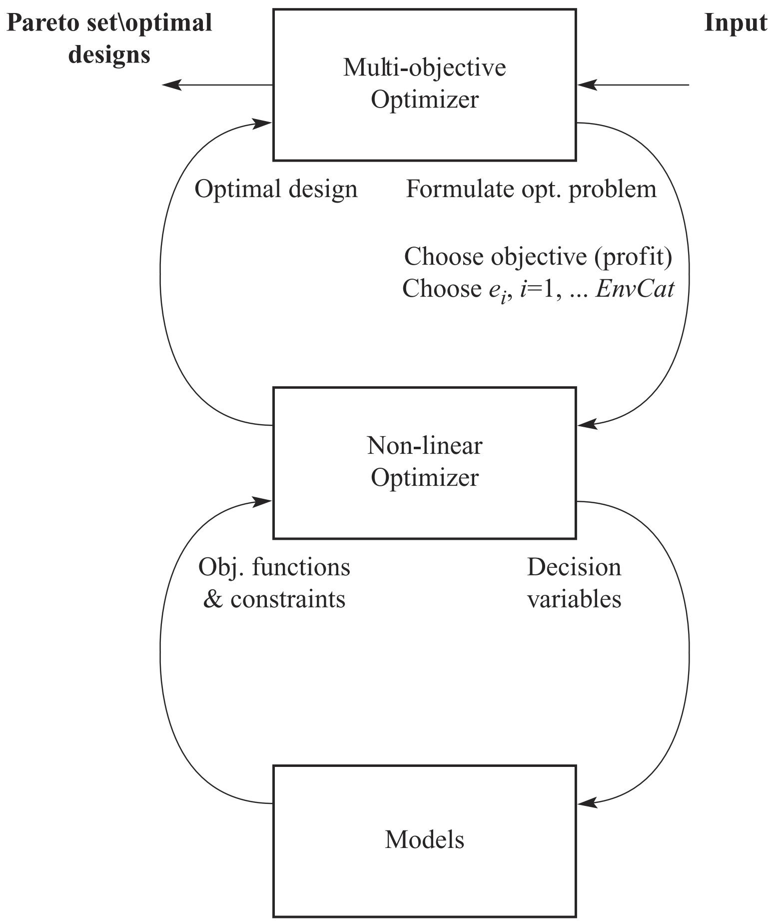


Figure 11.4 A generalized multi-objective optimization framework

constraints that will be systematically varied. In the middle loop a non-linear optimization program is called to search the decision variable (design) space for the optimal design, returning the value of the objective function and noting which constraints are met and the amount of slack for each, indicating which are binding. The non-linear optimizer requires multiple calls to the process simulation model in the inner loop of the algorithm. The overall search and bounding method is designed to search and map out efficiently the Pareto set of design alternatives. In the initial application of the method and the example that follows, only representative results are presented in terms of the bounds of the different objectives considered. In particular, the Pareto set is approximated by obtaining optimal designs with two values of ε_i ($i = 1, \ldots, EnvCat$ and $i =$ profit) for each objective function ($\dot{I}_i^{(NP)}$, $i = 1, \ldots, EnvCat$, and Profit). As such, as a first step in obtaining the overall Pareto set, each environmental objective is characterized in terms of its relative trade-off with respect to profit.

Benzene Production: a Benchmark Multi-objective Example

As the first step toward a multi-objective analysis with broad consideration of ecological protection, the benchmark problem of hydrodealkylation of toluene to form benzene is again considered here. The simulation model for this flowsheet was described earlier. The objective is to illustrate the benefits of using the multi-objective optimization framework to obtain alternatives with minimum environmental impacts and maximum profit.

The important control parameters – molar flow rate of the hydrogen feed, molar flow rate of the toluene feed, furnace temperature and conversion of the adiabatic reactor – are chosen as the decision variables for this multi-objective analysis. The different objectives include maximizing the annualized profit and minimizing the different environmental impacts of the output as calculated by the WAR algorithm, subject to the following process and product constraints:

- the benzene production rate must be maintained at 120kmol/hr;
- the adiabatic reactor must have a volume less than 500m^3;
- the hydrogen feed must have a purity of 95 per cent;
- the purity of the benzene product is at least 95 per cent.

The potential environmental impact indices for each of the chemicals present in the HDA process are listed in Table 11.4. The ozone depletion potential (ODP) indices and acid rain potential (ARP) indices for all components are zero, hence there is no need to include them as separate objectives. Furthermore, the indices for all components of human toxicity potential by ingestion (HTPI) and terrestrial toxicity potential (TTP) are equivalent, and the optimal solutions for minimizing or maximizing them are the same; hence they are listed together. The reduced problem thus includes six total objectives: HTPI or TTP, HTPE, ATP, GWP, PCOP and an economic objective. The economic objective is represented in terms of the annualized profit. The cost model (Diwekar *et al.* 1992) is represented by linear, fixed-charge costs. For details of this case study, please see Fu *et al.* (2000).

In the HDA process, benzene is the desired product and diphenyl, which is also formed during this process, can be either treated as a pollutant or sold as a by-product. These two cases are considered here.

Table 11.4 Potential environmental impact indices for the components in the HDA process

	HTPI	HTPE	ATP	TTP	GWP	ODP	PCOP	ARP
Hydrogen	0	0	0	0	0	0	0	0
Methane	0	0	0.057	0	0.0035	0	0.014	0
Benzene (product)	0	0	0	0	0	0	0	0
Toluene	0.078	2.2e-06	0.065	0.078	0	0	1.2	0
Diphenyl (pollutant)	0.12	0.0016	0.88	0.12	0	0	0	0
Diphenyl (by-product)	0	0	0	0	0	0	0	0

Case 1: diphenyl as a pollutant

Figure 11.5 shows the results of 10 different optimal designs for the HDA process by minimizing and maximizing each of the nine objective functions and removing the duplicate designs. These designs represent a first approximation to the complete Pareto surface consisting of many such designs. While this is only an initial exploration of the design space, the results in Figure 11.5 do show that profit does not conflict with the environmental criteria in all cases, and one can find an optimal design that is effective in meeting both economic and environmental objectives. This is due to the non-convex nature of the objective surface for the HDA process. From the figure, it can be seen that designs 1 and 2 are likely to be deemed superior to the others since (a) the profit of design no. 1 is the highest of all 10 designs and its environmental impacts (except for the value of PCOP) are lower than designs 3 to 10, and (b) design no. 2 has the best environmental performance (except PCOP), and its profit is only 7.4 per cent less than that of design no. 1. Further, the results

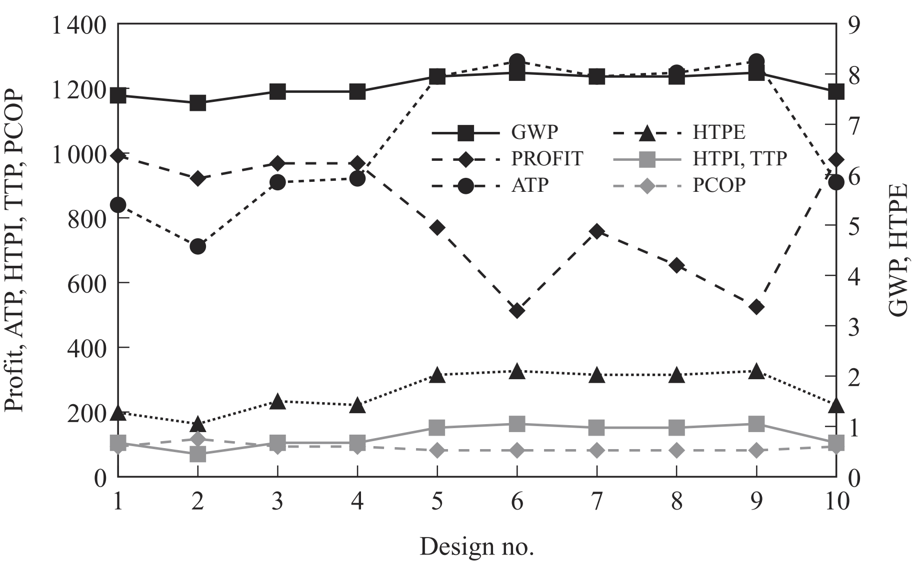


Figure 11.5 Approximate Pareto set for the HDA process multi-objective optimization (case 1: diphenyl as a pollutant)

indicate that designs 6 and 8 are the worst designs as they have the lowest profit and high environmental impacts.

Case 2: diphenyl as a by-product

Figure 11.6 presents the results of 10 different designs for the HDA process considering diphenyl as a by-product (that can be sold in the market). Again these results were obtained by minimizing and maximizing each of the six objective functions using the non-linear optimizer, and then removing the duplicate designs.

As in the previous case, profit follows a trend similar to that of a number of the projected environmental impacts: HTPI, TTP, PCOP and HTPE indicating once again the potential for designs that are both economically and environmentally attractive. However, the desirable and undesirable designs suggested by the analysis do differ from those derived for the previous case. When diphenyl is treated as a marketable product, rather than a pollutant, the undesirable designs can clearly be eliminated; however, the best designs are more difficult to identify. For example, designs 2, 6 and 8 in Figure 11.6 are likely to be eliminated when compared to design no.5, because designs 2, 6 and 8 have higher environmental impacts (along most dimensions, though a number of the environmental impact indices exhibit only a small amount of variation between designs, once diphenyl is removed from consideration as a pollutant) and lower profit than does design no. 5.

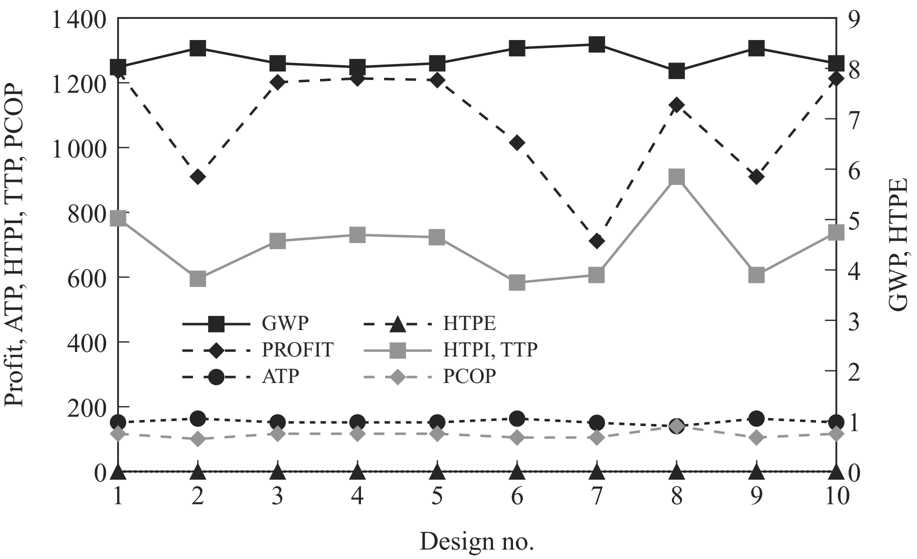


Figure 11.6 Approximate Pareto set for the HDA process multi-objective optimization (case 2: diphenyl as a by-product)

The multi-objective framework presented in this example helps to identify choices for the designer among the different economic and environmental objectives considered. In this case, we suggest that especially good designs will be those that have (a) higher profit and lower environmental impacts, (b) lower environmental impacts at the expense of small profit loss, (c) higher profit at the expense of slightly higher environmental impacts, and (d) lower environmental impacts for some categories at the expense of slightly higher environmental impacts in others. It is then a value judgment by the decision maker(s) to determine which design among these is the most appropriate.

The analysis presented thus far has been *deterministic*: relationships between the system design and economic and environmental performance are assumed to be known and modeled with certainty. In reality major uncertainties are usually present, and these can have a significant effect on the results. Methods for addressing uncertainty in process simulation, design and optimization are considered in the following section, as the next major challenge to implementing efficient, environmentally conscious process design.

A MULTI-OBJECTIVE OPTIMIZATION FRAMEWORK UNDER UNCERTAINTY

A probabilistic or stochastic modeling procedure involves (a) specifying the uncertainties in key input parameters in terms of probability distributions, (b) sampling the distribution of the specified parameter in an iterative fashion, (c) propagating the effects of uncertainties through the process flowsheets, and (d) applying statistical techniques to analyze the results (Diwekar and Rubin 1991).

As regards specifying uncertainty using probability distributions, to accommodate the diverse nature of uncertainty, different probability distribution functions can be used (Morgan and Henrion 1990; Taylor 1993; Cullen and Frey 1999). Some of the representative distributions are shown in Figure 11.7. The type of distribution chosen for an uncertain variable reflects the amount of information that is available. For example, the uniform and log uniform distributions represent an equal likelihood of a value lying anywhere within a specified range, on either a linear or a logarithmic scale, respectively. A normal (Gaussian) distribution reflects a symmetric, but decreasing, probability of a parameter value above or below its mean value. Normal distributions often result from the summation of multiple errors, and are often used to represent small measurement errors. In contrast, log normal distributions are positively skewed (with a heavy upper tail) and often result from multiplicative, order-of-magnitude variation or errors. Triangular distributions indicate a higher probability towards the mid-range of values, but may be specified to be symmetric, positively or negatively skewed. A beta distribution provides a wide range of shapes and is a very flexible means of representing variability over a fixed range. The standard beta distribution, for random variables between zero and one, is often used to represent uncertainty in a chemical mixture fraction or a product or process failure probability. Finally, in some special cases, empirical, user-specified distributions can be used to represent arbitrary characterizations of uncertainty (for example, fixed probabilities of discrete values based on observed samples).

Once probability distributions are assigned to the uncertain parameters, the next step is to sample the uncertain, multi-variable parameter domain. Alternatively, one can use collocation-based methods to derive a response surface of the actual uncertainty surface (Tatang 1994). Although this method can require significantly fewer runs than a sampling method, one needs to have substantial knowledge of the model, and discontinuities or

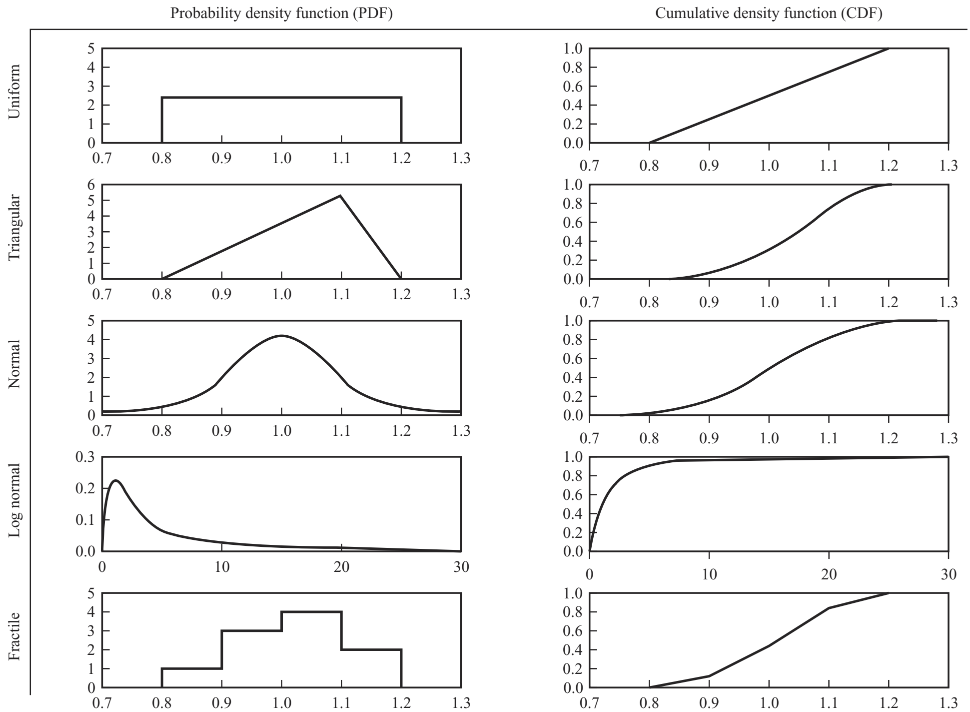


Figure 11.7 Probabilistic distribution functions for stochastic modeling

non-smoothness can result in erroneous results. Thus sampling methods provide the most generally applicable approach, and are discussed in more detail below.

One of the most widely used techniques for sampling from a probability distribution is the Monte Carlo method, which samples the parameter *in a purely random* manner, that is, all samples are independent and identically distributed from the overall target distribution. The main advantage of the Monte Carlo method is that the simulation results can be analyzed using classical methods of statistical estimation and inference. Nevertheless, in most applications, the actual relationship between successive points in a sample has no physical significance; hence the randomness/independence of successive samples used in approximating the target distribution is not critical. In such cases, constrained or stratified sampling techniques can allow better representation of the target distribution with a much smaller sample size.

Latin hypercube sampling is one form of stratified sampling that can yield more precise estimates of the distribution function. Here the range of each uncertain parameter X_i is sub-divided into non-overlapping intervals of equal probability. One value from each interval is selected at random with respect to the probability distribution within the interval. The n-values thus obtained for X_1 are paired in a random manner (that is, equally likely combinations) with n-values of X_2. These n-values are then combined with n-values of X_3 to form n-triplets, and so on, until n k-tuplets are formed. The main drawback of this stratification scheme is that, while it is uniform in one dimension, it does not ensure uniformity properties in k dimensions. Recently, an efficient sampling technique based on Hammersley points (Hammersley sequence sampling – HSS) has been developed (Diwekar and Kalagnanam 1997). This method uses an optimal design scheme for placing the n-points on a k-dimensional hypercube. This scheme ensures that the sample set is more representative of the population, showing uniformity properties in multi-dimensions, unlike Monte Carlo, Latin hypercube and its variant, the Median Latin hypercube sampling technique. It has been found that the HSS technique is at least three to 100 times more efficient than LHS and Monte Carlo techniques and hence is a preferred technique for uncertainty analysis, as well as optimization under uncertainty. Limitations in the effectiveness of methods with high uniformity (such as HSS) can occur when the uncertain parameters exhibit highly periodic properties or effects; however, such cases are expected to be unusual in most process design applications.

An Efficient Multi-objective Optimization under Uncertainty

The following is a brief description of an efficient multi-objective optimization framework under uncertainty based on the HSS technique. The details of the algorithms can be found in Fu and Diwekar (2001). Figure 11.8 shows the generalized framework of multi-objective optimization under uncertainty. Once again, as in Figure 11.4, the outer multi-objective optimization framework is used to formulate a number of optimization problems to generate optimal alternative solutions within the Pareto set. The innermost loop incorporates uncertainty by converting the deterministic model to a stochastic one. In the innermost loop, the HSS technique is used to generate distributions of uncertain parameters, which then map into the probability distribution of corresponding objective functions and constraints computed by the model. In the outermost loop the HSS technique is also employed to formulate combinations of right-hand sides for the constraint method so that the minimum number of optimization problems can be identified and solved to attain an accurate representation of the whole Pareto set. By using an efficient sampling technique for uncertainty analysis and for multi-objective optimization, and an efficient iteration between the optimizer and the sampling loop, this approach allows significant computational savings, bringing a number of real-world, large-scale problems that were previously unsolvable within reach for effective design and optimization. The case study for benzene production, used earlier to illustrate the usefulness of the multi-objective optimization framework, is now used to illustrate the effect of considering uncertainty in determining the solution surface.

Benzene Production: Uncertainty Results

As a first step, with the same HDA flowsheet, we ignore the uncertainties in economic and other input parameters. We again consider the case with diphenyl assumed to be a useful byproduct, so that the potential environmental impact values for the diphenyl stream are set to zero. To allow for uncertainty, all non-zero potential environmental impacts are assumed uncertain with the log-normal distributions shown in Table 11.5. The common standard deviation of the logs in each case implies that the uncertainty range (plus or minus three standard deviations) for each of the potential environmental impact values ranges from a factor of 10 below to a factor of 10 above the deterministic values assumed

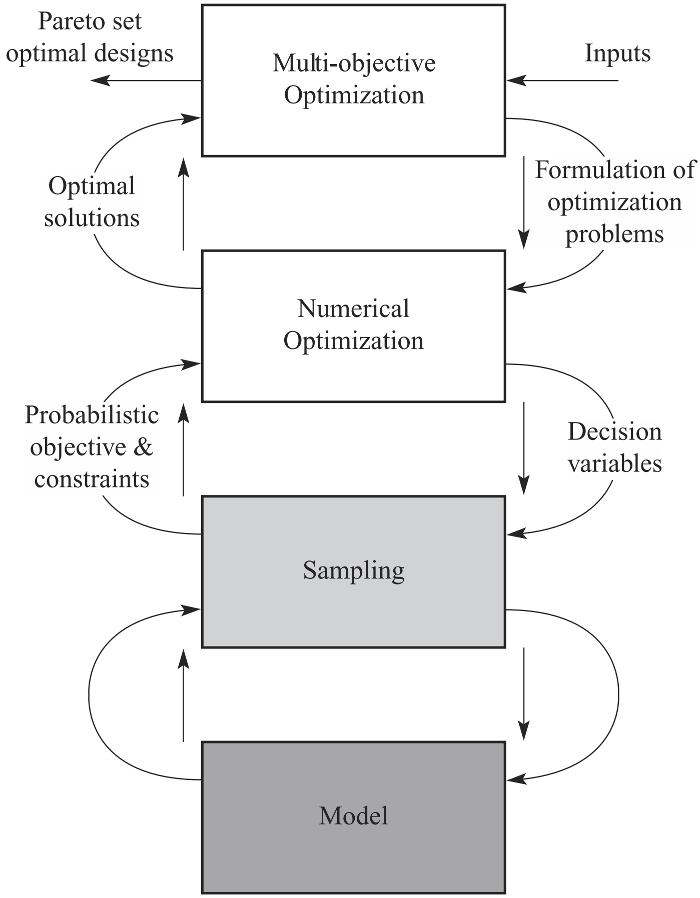


Figure 11.8 The multi-objective optimization under uncertainty framework

in Table 11.4. Two-order of magnitude uncertainty in environmental impacts is not uncommon, given the highly diverse and aggregate nature of the environmental indices considered.

Since there are no uncertainties factored into the economic objective, the same optimal design is obtained when only profit maximization is considered, as was obtained for the deterministic case. However, the results for the environmental objectives are no longer single values; rather they now follow probability distributions. The results for these are described by the cumulative distribution functions shown in Figure 11.9. Also shown in this figure (with a vertical line in each case) are the environmental impact index values determined for the previous, deterministic case. The results indicate that there is a 59 per cent probability that PCOP will be higher than the deterministic estimate shown by the horizontal line, and similarly 54 per cent, 52 per cent, 53 per cent and 51 per cent probabilities for HTPI, HTPE, ATP and GWP, respectively, being higher than the deterministic estimates when uncertainty factors in the potential environmental impacts are considered. While useful knowledge, this type of uncertainty characterization, after the

Table 11.5 Uncertainty quantification in environmental impacts indices for the components in the HDA process

	HTPI	HTPE	ATP	TTP	GWP	ODP	PCOP	ARP
Hydrogen	0	0	0	0	0	0	0	0
Methane	0	0	U3	0	U5	0	U6	0
Benzene (product)	0	0	0	0	0	0	0	0
Toluene	U1	U2	U4	U1	0	0	U7	0
Diphenyl (by-product)	0	0	0	0	0	0	0	0

Uncertainty factors	Type of distribution	Parameters (of natural log) Mean	Standard deviation
U1	Log normal	2.551	0.7675
U2	Log normal	−13.0271	0.7675
U3	Log normal	−2.7336	0.7675
U4	Log normal	−5.655	0.7675
U5	Log normal	−6.5713	0.7675
U6	Log normal	−4.2687	0.7675
U7	Log normal	0.1823	0.7675

design is specified, does little to indicate how explicit consideration of uncertainty may have redirected the design in the first place. For this, an integrated procedure, such as that show in Figure 11.8, is needed.

One approach for using the information on environmental impact uncertainty explicitly in the design optimization is to define a probabilistic objective function in terms of the mean (environmental impact and/or cost), the probability of exceeding certain values of these, the variance, or the median value of the objectives, depending on the decision maker's choice. For illustration purposes, we choose the mean value of each potential environmental impact to include as part of the objective function.

Figure 11.10 shows the different mean potential environmental impacts and profit for 10 optimal designs generated as an approximation to the Pareto set under uncertainty. The trends for the potential environmental impacts are similar to those determined for the deterministic case. This can be attributed to the fact that we have considered only the uncertainties in the environmental impacts for each component and these quantities are related to each environmental objective via a linear function. However, even from this first stage analysis, it is apparent that the relative effects of uncertainties on each objective function are different. This is illustrated in Figure 11.11, which demonstrates that, while the uncertainties in environmental impact have little impact on the profit, the mean environmental impacts are higher in the case where uncertainties are explicitly considered.

This benzene production case study is carried out entirely in the ASPEN simulator environment and provides a first step toward the process analysis approach to industrial ecology presented in this chapter. The major result is that environmental objectives need not conflict with economic benefits, as is often believed. This approach can be easily extended to industrial symbiosis. For example, Chang and Allen (1997) used multi-objective optimization combined with simplified material and energy balance models to identify chemical

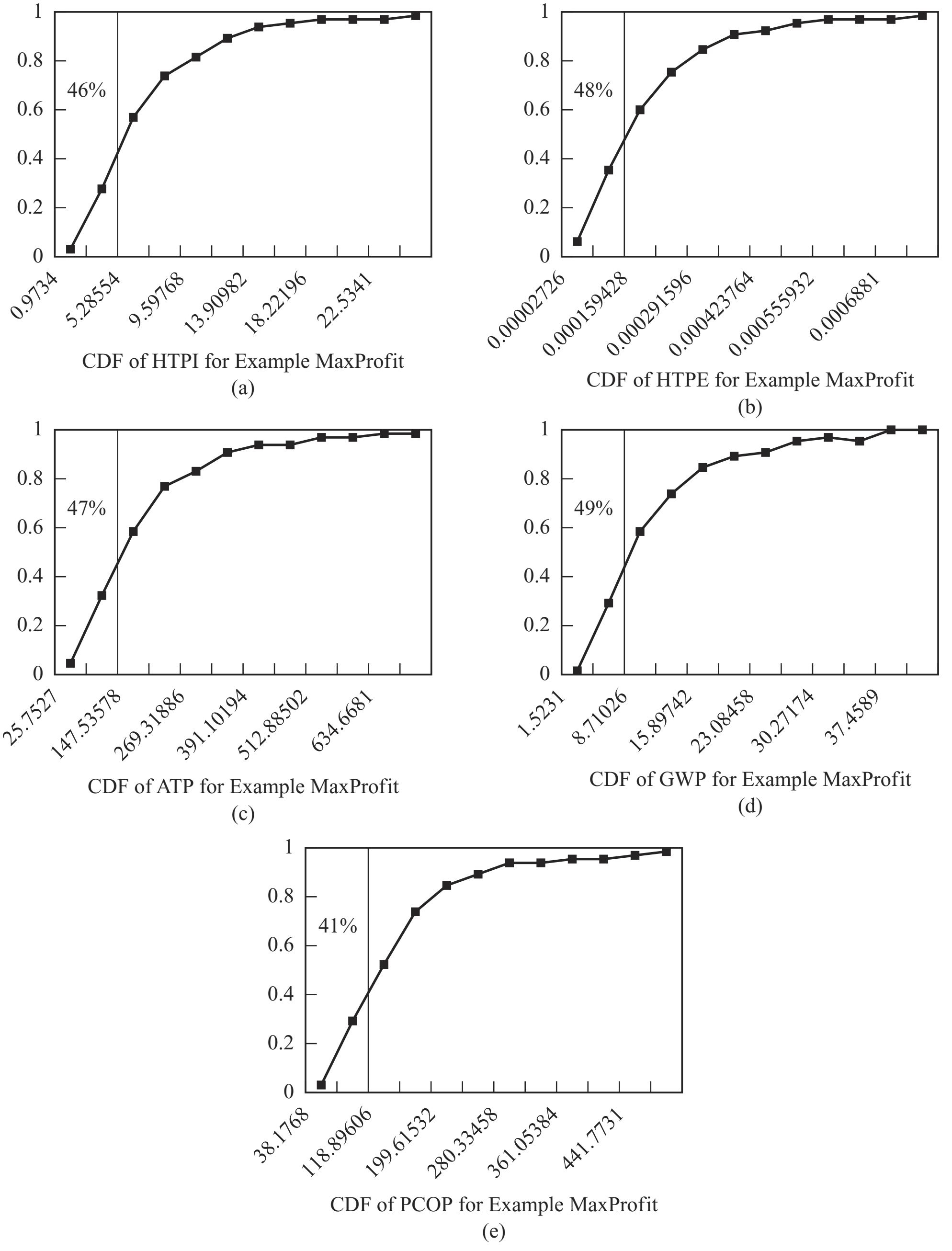


Figure 11.9 Uncertainty quantification in environmental impacts indices for the case study

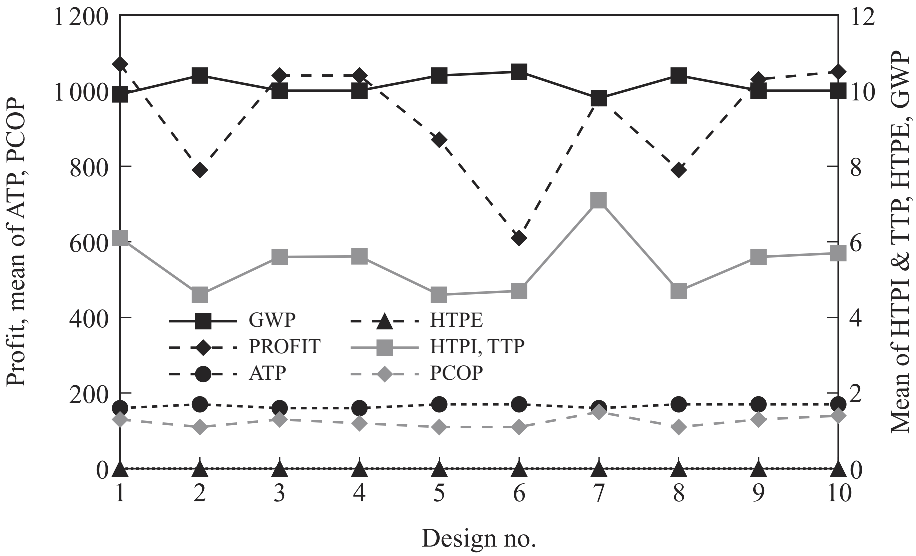


Figure 11.10 Approximation of Pareto set for the uncertainty case

manufacturing technologies for chlorine use for various industrial systems. However, to address the question of inaccuracies in the models, and lack of data, the problem of uncertainty (not considered by Chang and Allen) must be dealt with using methods such as those presented herein.

CONCLUSIONS

This chapter has presented a conceptual framework for a process analysis approach to industrial ecology. Current process simulation technology based on mass and energy balance principles can provide a unified framework for this approach. The capabilities of existing process simulation tools and their deficiencies in performing this task have been elucidated. A multi-objective optimization framework provides a mechanism to include the multiple, often conflicting, goals associated with industrial ecology. However, to address the issues of accuracy and relative weights assigned to these goals one must wrestle with the problem of uncertainty – in this case addressing how to value different environmental impacts, some of which are well characterized and some highly speculative. Uncertainty analysis coupled with the multi-objective framework can be truly beneficial in this context. This framework can also provide a basis for dealing with the problem of dispersed and scarce data, given that there is little or no commercial experience with industrial symbiosis, or with applying industrial ecology at larger scales, in practice. While the case study of benzene production illustrates the usefulness of the process analysis approach to industrial ecology using multi-objective optimization under uncertainty, we expect that applications at higher levels of economic aggregation, at the plant, community, national and even global level, will one day provide comparable insights into broader strategies for improving economic and environmental sustainability.

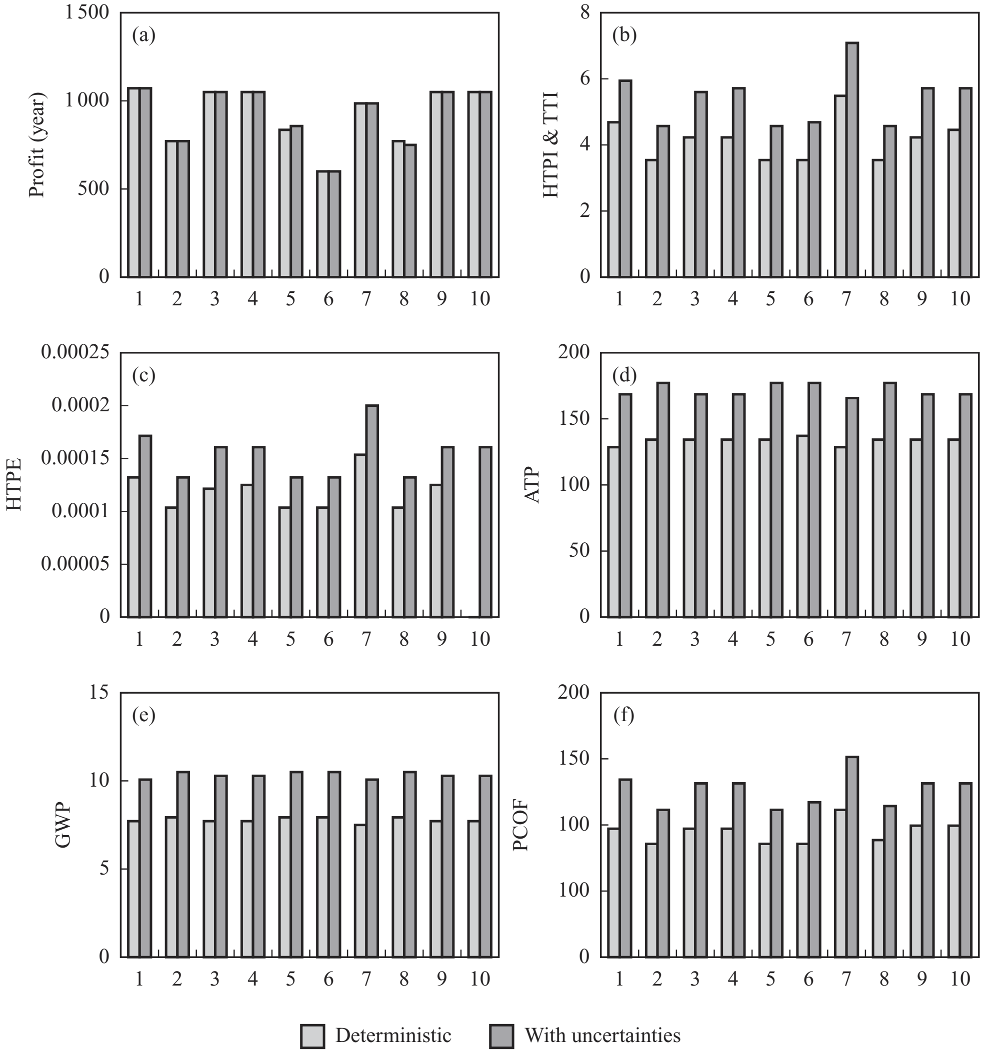


Figure 11.11 Relative effects of uncertainties on different objectives

12. Industrial ecology and life cycle assessment

Helias A. Udo de Haes

Life cycle assessment can be regarded as part of industrial ecology, which is a science that studies the interaction between society and its environment. In this field quite different approaches present themselves. First of all a distinction can be made between studies which are performed in physical terms and studies which are performed in monetary terms. Studies in physical terms have their historical roots in the 19th century and go back to Marx and Engels. These authors used the term 'metabolism' (*Stoffwechsel*) to imply a material relation between man and nature, a mutual interdependence beyond the widespread simple idea of man utilizing nature (cf. Fischer-Kowalski 1998). Studies in monetary terms may take the environment into account as physical extensions of monetary models, like input–output analysis as developed in the 1980s (Leontief 1986), or they may even address the environmental consequences of economic activities in monetary terms, as in cost–benefit analysis. The present chapter only includes studies of the society–environment relationship in physical terms.

In this field of physical relationships a further distinction can be made regarding different types of object. Thus environmental risk assessment (ERA) has its focus on the assessment of environmental impacts of single activities like the functioning of a factory, or of single substances. In fact ERA studies start with the emissions and do not really consider the processes in the economy which precede them. Then there are studies which have their basis in physical equilibrium models of energy, materials or substances within both society and its environment. The basis for these studies has been laid down by Ayres and Kneese (1969). They have resulted in tools like energy analysis, material flow accounting (MFA) and substance flow analysis (SFA). For a given area and for a given year the metabolism of these different flows is investigated, generally supposing an equilibrium situation, but now also extending towards dynamic modeling. And then there is a third approach with a focus on products, or more precisely on product systems; that is, the total of processes in the economy which are responsible for fulfilling a certain function. Here we are in the field of life cycle assessment, which will be the main subject of this chapter. Although mass balance principles can be of use here for broad checks, life cycle assessment goes beyond that; for instance, the production of dioxins in the incineration of waste cannot be traced in that way.

A SHORT HISTORY OF LIFE CYCLE ASSESSMENT

Life cycle assessment (LCA) originated in the early 1970s. In this initial period studies were performed in a number of countries, in particular Sweden (Sundström 1973), the UK (Boustead 1972), Switzerland (Basler and Hofmann 1974) and the USA (Hunt *et al.*

1974). The basis lay in energy and waste management problems; the products which got primary attention were beverage containers, a topic which had dominated the LCA discussions for a long time. During the 1970s and the 1980s numerous studies were performed, using different methods and without a common theoretical framework. The consequences were rather negative, because LCA was directly applied in practice by firms in order to substantiate market claims. The obtained results differed greatly, although the objects of the study were often the same, thus preventing LCA from becoming a more generally accepted and applied analytical tool.

Since about 1990, exchanges between LCA experts have increased. Under the coordination of the Society of Environmental Toxicology and Chemistry (SETAC) efforts started to harmonize the methodology (cf. Consoli *et al.* 1993), laying the basis for LCA as a broadly accepted formal tool. Since 1994, the International Organization for Standardization (ISO) has played a crucial role in this field; as also, since 1995, has the United Nations Environmental Programme (UNEP, Paris). Whereas SETAC has primarily a scientific task, focused on methodology development, ISO has taken up the formal task of methodology standardization, leading to the present standards in the 14040 series. UNEP has its focus on the global use of LCA.

As the LCA methods are becoming more sophisticated, software and databases are also being developed. However, for the credibility of the results procedural requirements are essential. Thus there generally will be a great need for an input by the most important stakeholders in the process, and there will be a need for an independent peer review of the results of an LCA study.

DEFINITION AND APPLICATIONS

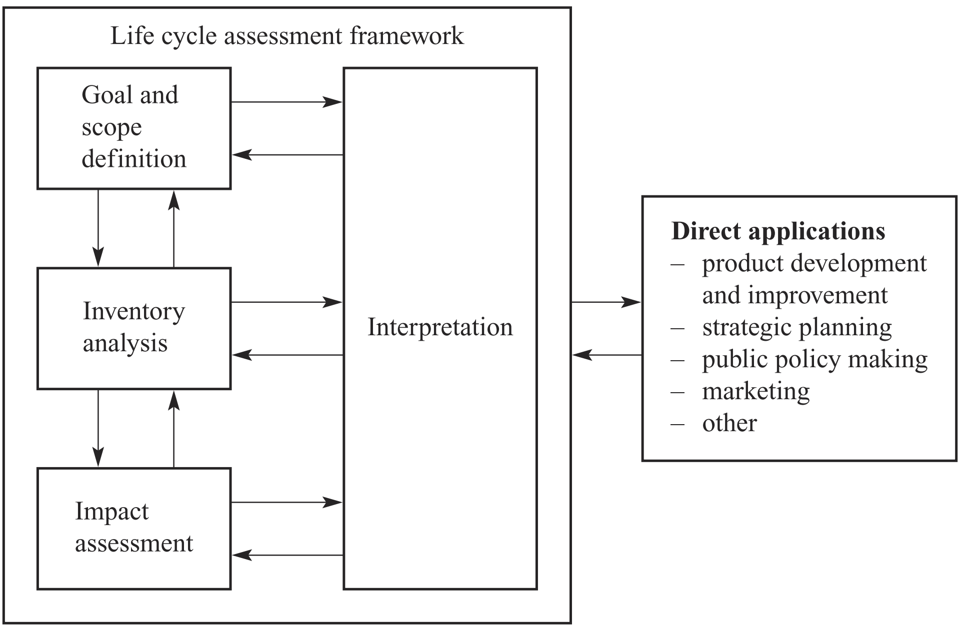
In ISO 14040, LCA is defined as follows: 'LCA is a technique for assessing the environmental aspects and potential impacts associated with a product by compiling an inventory of relevant inputs and outputs of a system; evaluating the potential environmental impacts associated with those inputs and outputs; and interpreting the results of the inventory and impact phases in relation to the objectives of the study.' Products also include services which provide a given function. In the following we, however, will speak of a product as *pars pro toto* for all objects of LCA, if not specified differently.

The reference for the study is the function which is delivered by a product. This means that ultimately all environmental impacts are related to this function, being the basis for comparisons to be made. The product, which delivers this function, is studied during its whole life cycle; all processes related to the product during its whole life cycle are together called the 'product system'. These processes are studied employing a quantitative, formalized mathematical approach. A clear distinction is made between objective and normative parts, thereby ensuring transparency.

LCA is applied at various levels, ranging from operation to strategic applications. It is used in operational management, including purchasing decisions; in communication and marketing, including the underpinning of ecolabeling programs; in product design and development contributing to the area of Design for the Environment; in the underpinning of capital investments; and in strategic planning (cf. Wrisberg *et al.*, 1997). The focus of applications is on large companies, but it increasingly includes governmental agencies and branch organizations of smaller companies. Whereas the ecolabeling programs in general have not met their expectations, use in the other types of applications shows a consistent increase over recent years (Frankl and Rubik 2000).

TECHNICAL FRAMEWORK

In order to make LCA a tool for comparative purposes a first step concerns standardization of a technical framework and of terminology. The ISO framework consists of the following phases: goal and scope definition, life cycle inventory analysis, life cycle impact assessment and life cycle interpretation (see Figure 12.1). From the figure it is apparent that LCA is not a linear process, starting with the first phase and ending with the last. Instead it follows an iterative procedure, in which the level of detail is subsequently increased.



Source: ISO (1996).

Figure 12.1 Technical framework for life cycle assessment

The *goal and scope definition* phase starts with a specification of the purpose and scope of the study. This includes the choice of the products which will and which will not be taken into account, which can be a point of serious debate. Further, the functional unit has to be defined. The functional unit is the central, quantitative measure of the function to be delivered. All processes to be investigated in LCA are to be related quantitatively to this functional unit. This first phase of the framework also includes the definition of the level of detail required for the application at hand, and the establishment of a procedure for ensuring the quality of the study.

The *inventory analysis* phase is the most objective and also the most time-consuming part of the study. It starts with the drawing of a flow chart of the processes involved in the product system, with their material and energy relationships. The use of the product is the central element; starting from here, the processes ramify 'upstream' through the production processes and up to the different resources used, and 'downstream' to the different ways of waste management involved. For a flow chart also the system boundaries have to be defined between the product system (as part of the economy) and the environment. This implies that the flows across these boundaries, the environmental interventions, have to be defined (for different options in this respect, see Figure 12.2). Another aspect of the definition of system boundaries concerns the demarcation in space and time. Here there is another difference between LCA and ERA and SFA. In the two latter, processes within a given region, and within a given period of time, are included. In contrast, in LCA as a holistic tool, the viewpoint is full integration over both space and time, generally without further specification of areas or time periods.

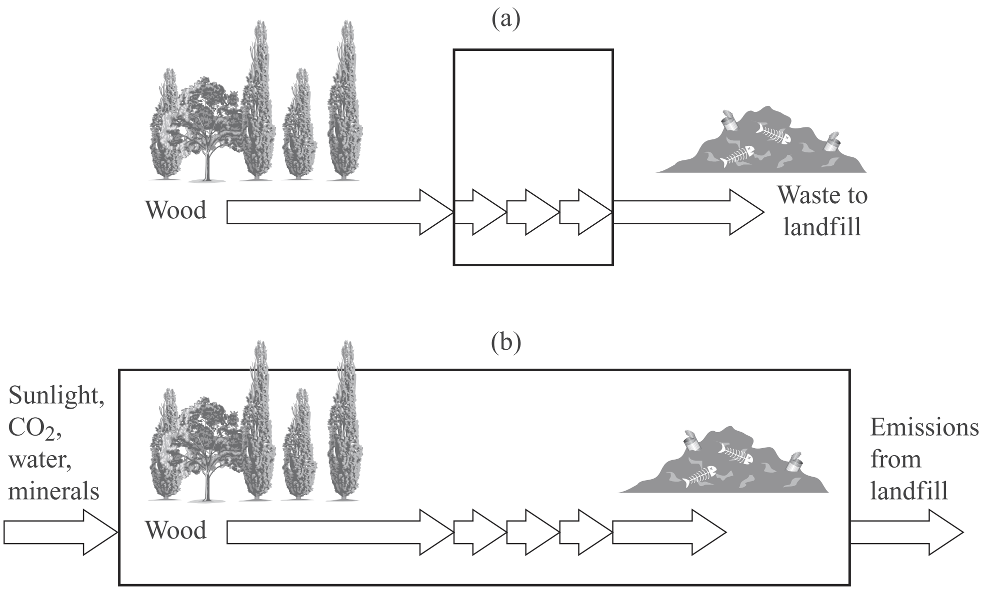


Figure 12.2 Two ways of defining system boundaries between physical economy and environment in LCA: (a) with narrow system boundaries, (b) with extended boundaries

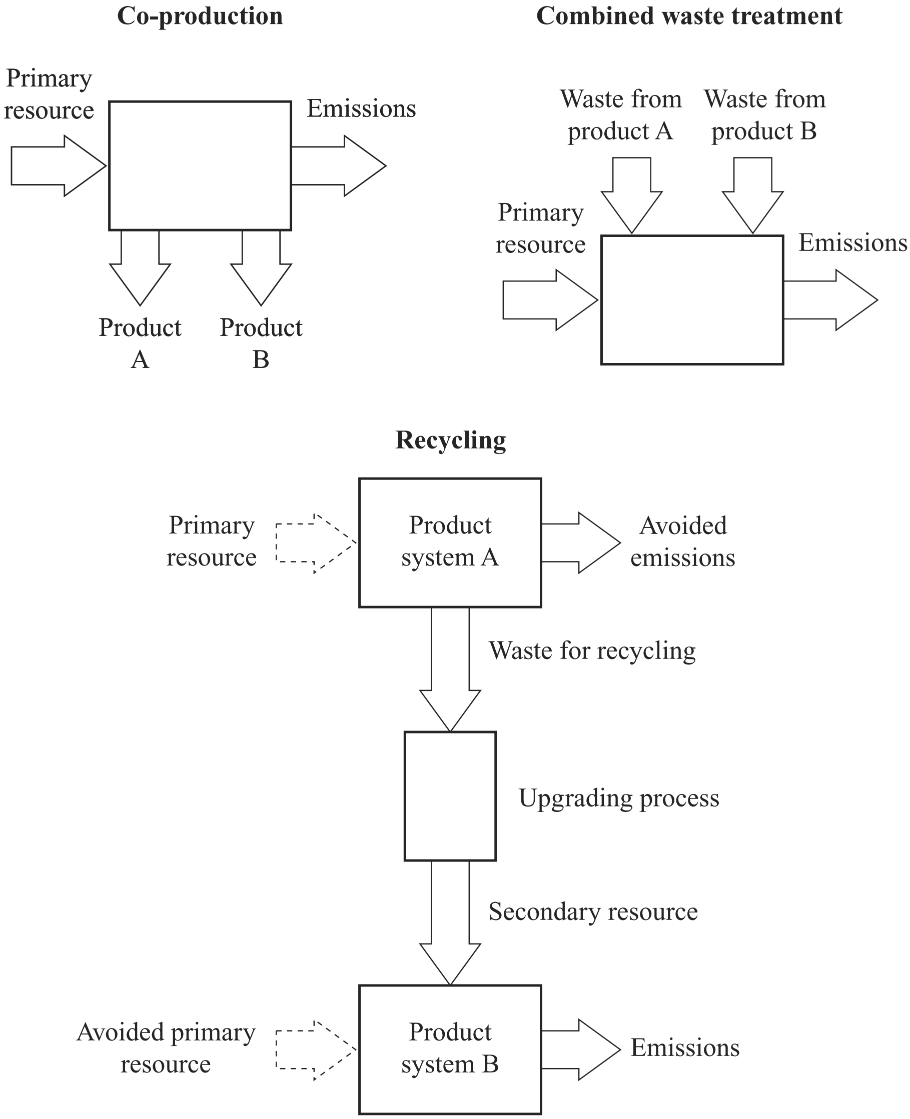
Given the system boundaries with the environment, the next step concerns the specification of processes which will be analyzed and those which will be left out. A recent development concerns a distinction between foreground and background processes, the former being analyzed in the usual detailed way, the latter being approached using input–output analysis as approximation (Hendrickson *et al.* 1998). The next step concerns the construction of the model and the gathering of data about the different inputs and outputs of the processes. The model must quantitatively relate the different processes to each other, using the magnitude of the functional unit as reference. A fundamental difference from ERA and SFA is that in LCA processes are included to the extent that they contribute to the defined functional unit; in contrast, in ERA and SFA they are always taken into account to their full magnitude.

A specific issue regarding the construction of the inventory model concerns the so-called 'multiple processes'; that is, processes which provide more than one function. Main examples are co-production, meaning that one unit process produces more than one product, combined waste treatment and recycling. In LCA this is called the 'allocation procedure' (see Figure 12.3). A general framework for allocation is developed in the ISO standard (ISO 14041). However, this still permits different calculation procedures based either on physical characteristics as the guiding principle (Azapagic and Clift 1999, 2000), on system extension (Weidema, 2001) or on economic principles (Huppes 1993). A more detailed harmonized set of rules is an important aim for future life cycle inventory development. The inventory analysis concludes with the compilation of the inventory table, the total list of the extractions and emissions connected with the product systems investigated. If a study is only performed up to the inventory analysis, it is called an LCI, that is, a life cycle inventory study.

The next phase concerns *life cycle impact assessment*, or LCIA. This phase interprets the extractions and emissions of the inventory table in terms of environmental issues, and it aggregates these data for practical reasons; a list of 50 or 100 entries cannot be dealt with in decision making. In the 1970s impact assessment was in fact done in an implicit way, by defining a number of broad, inventory-based parameters, which were thought to be indicative for the total spectrum of impacts. Examples of such parameters include net energy consumption, total input of resources and the total solid waste output (Hunt *et al.* 1974). A more recent example of this approach concerns the MIPS method (material input per service unit; Schmidt-Bleek 1993a, 1993b), in which the total material input of a product system is quantified. These approaches are time-efficient, and can lead to robust results. However, such a small number of inventory-based indicators is not very discriminatory and neglects various types of impact.

Since the mid-1980s, different methods for aggregating substances into a surveyable number of categories have been in development. Here guidance is being given by the ISO standard 14042. In this standard a stepwise procedure is defined that separates the scientific and the normative (that is, value-based) steps as much as possible. A number of impact categories are defined, together with the underlying characterization models; that is, the models for the aggregation of the extractions and emissions within the given impact categories. Here generally a 'problem theme approach' is followed, as originally proposed by CML in the Netherlands (Heijungs *et al.* 1992). The categories are defined as much as possible on the basis of resemblance in the underlying environmental processes, for instance all substances leading to an increase in infrared absorption and thus to possible climate change. But, clearly, value choices also are involved in characterization modeling (Owens 1997). Table 12.1 presents a list of impact categories, developed by a working group of SETAC Europe, as a structure for the analysis of the impacts and as a checklist for the completeness of the different types of impacts to be considered. A main distinction is made between input-related categories ('resource depletion') and output-related categories ('pollution').

The impact assessment phase also includes a number of optional steps. One of these concerns normalization, which involves a division of the results by a reference value for each of the impact categories, for instance the total magnitude of that category for the



Note: Horizontal arrows indicate flows from and to the environment; vertical arrows indicate flows from and to other product systems.

Figure 12.3 Allocation of environmental burdens in multiple processes

given area and moment in time. Thus the relative contribution to the different impact categories can be calculated, owing to the given product system. Another concerns weighting, being a formalized quantitative procedure for aggregation across impact categories, resulting in one environmental index. Such environmental indices are very practical to use, particularly in the ecodesign of products; they enable a fast comparison between materials which all have their environmental characteristics expressed in one single number.

Table 12.1 Impact categories for life cycle impact assessment

A. Input-related categories ('resource depletion')	
1. extraction of abiotic resources	glob
deposits such as fossil fuels and mineral ores	
funds such as groundwater, sand and clay	
2. extraction of biotic resources (funds)	glob
3. land use	loc
increase of land competition	
degradation of life support functions	
biodiversity degradation due to land use	
B. Output-related categories ('pollution')	
4. climate change	glob
5. stratospheric ozone depletion	glob
6. human toxicity (incl. radiation and fine dust)	glob/cont/reg/loc
7. ecotoxicity	glob/cont/reg/loc
8. photo-oxidant formation	cont/reg/loc
9. acidification	cont/reg/loc
10. nutrification (incl. BOD and heat)	cont/reg/loc
Flows not followed up to system boundary	
input-related (energy, materials, plantation wood, etc.)	
output-related (solid waste, etc.)	

Note: glob = global; cont = continental; reg = regional; loc = local.

Source: Based on Udo de Haes *et al.* (1999).

The last phase of LCA, according to ISO, is *life cycle interpretation*. During this phase, the results are related to the goal of the study as defined in the beginning. This includes the performance of sensitivity analyses and a general appraisal. A sensitivity analysis is of great importance for checking the reliability of the results of the LCA study with regard to data uncertainties and methodological choices. This can also lead to a new run of data gathering if the goal of the study appears not to be reached satisfactorily.

The next two sections will discuss two aspects which are relevant for usefulness of analytical tools like LCA. These are the choice of the model parameters, and the different ways to deal with uncertainty.

THE CHOICE OF THE MODEL PARAMETERS

The choice of the model parameters is not a technical matter only. Firstly, there is the predominating choice between modeling in terms of physical and of monetary parameters. This has been a point of debate since the 1970s. Modeling in physical parameters as in LCA or ERA stays closer to reality it does not make the assumption that all valuable objects can be expressed in market terms. Expressing human life in monetary terms is often perceived as a degradation of that value. On the other hand, use of monetary units can be seen as the ultimate step in aggregation. If all environmental issues can be expressed in one single index, such as after weighting in LCA, it is only one step further to put them in monetary terms, further simplifying decision making. But of course it is not as simple as that. The major problem is that some environmental issues can more readily be put in market terms than others. Damage to man-made resources can be monetized rather easily; but damage to human health involves many assumptions, and damage to ecosystems can hardly be covered in this way.

If choosing physical parameters, there is a comparable bias related to the types of impacts which are generally taken into account. In principle, LCA aims to be all-encompassing with respect to the types of impact to be analyzed. In practice, however, there is a focus on the extraction of resources and on emissions. It appears to be difficult to relate changes in land use to a functional unit. Consequently, land use changes are often neglected in LCA, although land use may be by far the most important factor affecting biodiversity. Such a restriction, be it methodology-driven or not, is encountered more often. Thus many policy analytical studies just take CO_2 as the main indicator of environmental burden, omitting all other types of impact. This can only be acceptable as long as the conclusions are also viewed in this limited context.

A third point related to the choice of the model parameters concerns the level of the cause–effect network at which they should be defined. This is a core issue in LCA, but may also play a role in a tool like ERA. Changes at early levels in environmental cause–effect networks, such as changes in climate forcing caused by greenhouse gases, or changes in proton release as caused by acidifying substances, can be assessed with relative high certainty. Moreover, these changes will be relevant for broad groups of impact, ramifying along subsequent environmental pathways. On the other hand, one can aim to define the modeling outputs at the so-called 'damage level'; that is, damage to human health, to ecosystems, to crops, to man-made materials or to works of art (for example, Goedkoop *et al.* 1998; Spadaro and Rabl 1999). Assessment of impacts at this level will generally be subject to high uncertainty, and will generally only be feasible for a small selection of all possible impacts involved; but the results will be much better understood, as they deal with the entities which are of direct concern to us (Udo de Haes *et al.* 1999).

HOW TO DEAL WITH UNCERTAINTY?

The use of analytical tools will generally involve many uncertainties. These can be technical uncertainties regarding data, they can be methodological assumptions, and they can be value choices or even paradigmatical differences. There are a number of options to deal with such uncertainties. Below we will briefly discuss the most important ones.

New Measurements

The most straightforward answer to uncertainties consists of new measurements. These can pertain to new dose-response experiments in the laboratory, to the validation of extrapolations from laboratory to field, or to the validation of field models like the multimedia dispersion models. This is the high road of uncertainty abatement. But it is time and money-consuming and will be no option for a given practical case study.

The Choice of Robust Indicators

A next possibility is to choose indicators which are rather robust. However, the choice of more robust (more certain) indicators will come at the cost of accuracy. For example, the impacts of chlorine policies may be assessed in terms of impact category indicators, like those used in LCA. These have a rather high resolution power, but many of them are quite uncertain. In contrast, these policies can be assessed in terms of total kilograms of chlorine emitted, as is generally done in SFA studies. Such a metric is very robust and may therefore arouse significantly less resistance in a policy debate (Tukker 1998). But very important differences between the emitted substances will then be obscured. Going even one step further, one may leave quantification altogether and choose qualitative indicators like 'made from recycled material' or 'biodegradable'. This may further reduce public resistance to the results, but will again be less informative.

Uncertainty Analysis, Sensitivity Analysis and Scenario Analysis

Given a set of indicators, the uncertainty thereof can be assessed in terms of standard errors. These errors will depend on many links in the chain of processes underlying the indicator at hand. Furthermore, the errors will pertain to uncertainty in data, to methodological assumptions or to value choices regarding these different links. Consequently, the results of uncertainty analyses will soon become very complex and may well pile up uncertainty upon uncertainty. A more sophisticated approach concerns Monte Carlo simulation. For every element in the uncertainty of an indicator the probability of different possible values is assessed. Then subsequent computation runs are made, in which the different uncertainty elements are fixed independently, each according to its own probability distribution. The final result will show a more realistic range of outcomes, which will avoid artificial accumulation of uncertainties.

If no uncertainty values can be given, a sensitivity analysis can be performed starting from deliberate changes in the modeling conditions. Thus changes which are deemed reasonable can be made in the input data, in the methodological assumptions or in value choices underlying the different steps in the methodology. The consequences of such changes for the final result can then be calculated. This procedure is used quite often, as it puts rather low requirements on study resources and still provides important insights in the robustness of the final results.

Sensitivity analysis is generally performed for separate parameters, regarding data, methods or value choices. In scenario analyses sets of choices are put together into consistent packages. Thus we can calculate a worst case, a most likely or a best case scenario. Scenario analyses thus help to structure the results of sensitivity analyses in order to make them more comprehensible for decision making purposes.

International Harmonization

International standardization in the field of analytical tools predominantly focuses on terminology, on technical frameworks and on procedural requirements. But it may also go one step further, in harmonizing the use of best available data or methods. Thus the Intergovernmental Panel on Climate Change (IPCC) working under UNEP authority, among others, establishes the best available knowledge about climate change due to different greenhouse gases in terms of the well-known global warming potentials (GWPs). Likewise, the World Meteorological Organization (WMO) establishes best values for the stratospheric ozone depletion potential (ODP) of different substances. Recently, a combined research program has been defined by SETAC and UNEP to identify best available practice also for other impact categories. Although considerable uncertainties may be involved, such harmonization guides practical application and helps to avoid arbitrariness in selecting best data or models.

Procedural Checks

The above options for dealing with uncertainty all regard technical characteristics. Quite another approach starts from the other side, that is, from the decision procedure in which the results of the analytical tools are to be used. For instance, the results can be reviewed by an independent panel of experts, or even by a panel of stakeholders. If the results pass such a review procedure, this may well contribute more to the credibility of the results than any of the above technical procedures. For this reason, much attention is currently paid to the possibilities of incorporating analytical tools like LCA in explicit decision procedures in which both independent experts and the relevant stakeholders have a clearly defined input. An example concerns a European directive which gives guidance on the acceptability of the type of packaging to be used (when a company is allowed to use non-reusable materials); or a directive which guides the choice between waste management options.

Paradigmatic Differences

The most fundamental problem can be that analytical tools involve paradigmatic assumptions which are not shared by the different stakeholders in a decision process. Thus there is a major gap between a risk approach, as used in tools like LCA and ERA, focusing on emissions which actually take place, and a precautionary approach, focusing on inherent risks of a process. Such a gap cannot be bridged by improving the models or data used, or by better public participation in the decision process. Such differences can lead to grave frustrations regarding the application of quantitative analytical tools like LCA or ERA. Examples are the historic public debate on the acceptability of nuclear power installations, the debate on the environmental risks of the chlorine industry and materials like PVC (Tukker 1998), and more recently on the use of genetically modified organisms (GMOs). Generally one will have to go back to the precise questions being asked and to the way risks are approached. The use of quantitative analytical tools presupposes agreement on these points.

CONCLUSIONS

Life cycle assessment (LCA) concerns one of the major approaches in the field of industrial ecology. It involves a cradle-to-grave analysis of product systems, that is, of the total of processes which are involved in the provision of a certain function. It is complementary to other tools, such as environmental risk assessment, focusing on the environmental impacts of single activities or single substances, or substance flow analysis, focusing on the metabolism of substances in the economy as well as in the environment. LCA is a formal, quantitative tool in the area of LCA. Main contributing organizations are SETAC, responsible for its scientific development, ISO, responsible for its international standardization, and UNEP, taking a leading position in the enhancement of its global use. LCA appears to be increasingly used by industry, from operational decisions, like the purchasing of materials, up to strategic decisions. Like other formal analytical tools, LCA has a number of clear limitations. Some of these can be tackled by technical measures, some by procedural measures. But some limitations deal with paradigmatic differences regarding the way one wants to cope with risks. Decision procedures involving stakeholders with a risk approach versus stakeholders with a precautionary approach cannot easily be supported by LCA or other formal and quantitative environmental assessment tools.

13. Impact evaluation in industrial ecology

Bengt Steen

The focus of this chapter is on evaluation of impacts from emissions, resource extractions and other interventions from human activities and technical systems on our environment. The analysis of technical systems is only briefly touched upon. The term 'evaluation' is used to represent a subjective view on descriptions of processes and states in objective physical terms. This means that both physical parameters and human attitudes and preferences are included.

Evaluation of environmental impacts from human activities is made in several contexts in society and several evaluation methodologies or methodological frameworks exist. Sometimes these are called 'tools' and thought of as being part of a 'toolbox'. When needed, the appropriate tool is picked out of the toolbox and used for impact evaluation. In reality the flexibility of the various tools is such that they overlap in many applications. The tools have many similarities but their focus and terminology vary.

The oldest tool is probably *risk assessment* (RA). There are three types of risk assessment: for human health, for ecological health and for accidents. The first two are mostly used for chemicals and the third for industrial activities (including chemical manufacturing). They all are carried out in a similar way: first there is a hazard identification step, second there is a risk estimation step (hazard impact times probability) and third there is a risk communication step. The RA methodology has been strongly influenced by several major industrial accidents (notably Seveso and Bhopal) though it has evolved in the direction of assessing risks of chemicals being introduced or used in a market context. The information gained from risk assessment is intended to be used to support a decision about issuing a permit or for formulating rules or restrictions about its use. In short, risk assessment aims at identifying risks and decreasing them to an acceptable level.

A similar aim lies behind environmental impact assessment (EIA), but the object of study is usually not a chemical substance but the building and operation of an industrial plant or other large-scale technological projects. Compared to the situation when making a risk assessment, there is a known location of the activity and the amounts of various substances involved are fairly well known. An EIA therefore involves compiling an inventory and description of the surroundings and dispersion modeling of emissions from the plant.

Measures to reduce environmental impacts are often evaluated by economic techniques. The best-known technique is called *cost–benefit analysis* (CBA). CBA analysis may vary much in depth with regard to the impact evaluation. Sometimes the economic value of an impact is determined by just asking people how much they are willing to pay (known as WTP) to reduce the pollutant concentrations by some amount (say, by half). There is a symmetric technique, called willingness to accept (WTA) in which people are asked how much they would be willing to accept in exchange for some defined reduction in environmental amenity. A rather different approach, known as 'hedonic' analysis, attempts to account for the price or asset value of a complex good, by disaggregation into contributions from different attributes (Pearce 1993; Herriges and Kling 1999). Thus real estate values in otherwise similar areas may reveal an implicit valuation for (avoiding) sulfur dioxide pollution from a nearby plant. Sometimes the valuation modeling is more elaborated, as in the ExternE project of the European Commission (1995). Other techniques include environmental cost accounting, environmental accounting and life cycle costing (LCC). All these may be included under the rubric of 'environmental economics'.

In life cycle assessment (see Chapter 12.5) and its subprocedure life cycle impact assessment (LCIA), the object of study is a product or service. The goal of an LCA may vary but, mostly, the LCIA is intended to be used – sooner or later – in a choice between two products or processes. The comparative element in LCIA requires a comprehensive approach, in which the focus is different from risk assessment and EIA. Besides studying each impact type separately, the weighting of various impacts becomes an issue. The LCIA procedure is standardized by the International Standards Organization (ISO) and described in the ISO 14042 standard. A technical report (ISO TR 14047) is at present being worked out with examples on how the standard may be implemented (ISO 2000).

Engineering science and natural science make different demands on LCIA. In engineering science the product's overall performance is the focus. The product is intended to function as well as possible in a number of situations. In natural science, the theory is central. The theory is intended to function as well as possible in a number of situations. The inclusion of uncertain models or data in a natural science-oriented context may be objectionable, whereas omission of it would be objectionable to the engineering scientist, as it would be tantamount to neglecting a likely problem. Experience, in particular from the LCA area, has revealed many such methodological conflicts.

From a system analysis point of view, all impact evaluation techniques may be seen to deal with the technical, natural and the social subsystems (Figure 13.1). The technical system may be further divided into a foreground and a background system. The foreground system is the one you know and can specify in detail. The background system includes, for instance, market behavior and infrastructure.

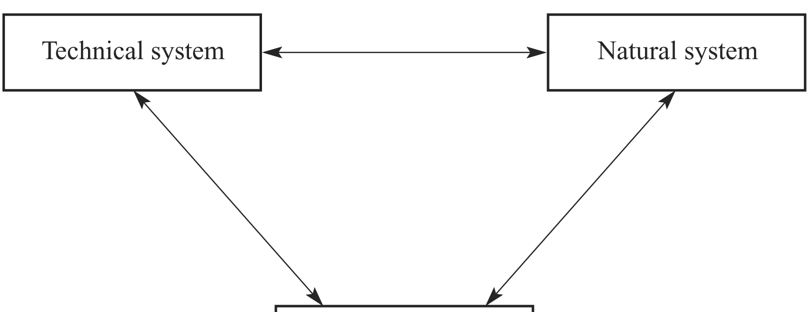


Figure 13.1 An impact evaluation combining scenarios for technique, environment and human attitudes

In industrial ecology (IE) you will need a toolbox for different types of impact evaluations. But instead of describing the different tools as they mostly are used, one by one, some procedural steps, which are common to all tools, will be used to structure the text of this chapter:

- formulation of goal and scope;
- selection of impact indicators;
- modeling or recognizing interactions between technical system indicators and impact indicators;
- comparing different types of impacts and evaluation of total impact;
- analysis of uncertainty and sensitivity;
- data documentation and reporting.

FORMULATION OF GOAL AND SCOPE

The choice of goal and scope has a very significant influence on the outcome of an impact evaluation. Experience shows that this rather obvious statement needs to be repeated often. There are numerous examples of misunderstandings arising when telling an impact evaluation story without adequately specifying the goal and scope.

The choice of goal and scope is an ethical or normative issue that is normally left out of discussion in a scientific context. However, in the ISO documentation on LCA standards (ISO 2000) it is recommended that the impact evaluation should include all significant impacts on human health, ecosystems and natural resources. Generally speaking, there are three questions that need to be addressed when setting out a goal and scope: What is to be included in the study? How to deal with trade-offs? How to handle uncertainty?

When deciding upon what to include in the study, there are many dimensions to keep in mind. One is the *qualitative* dimension. In general terms, one may think of things to include as belonging to 'safeguard subjects' or 'areas of protection', such as human health or natural resources. In LCA the concept of impact categories exists, which is more focused but still not a quantitative indicator. The quantitative indicators, called 'category indicators' in LCA and 'impact indicators' in many other methodologies, define the qualitative system borders of the 'environment' we study.

Another dimension where system borders need to be set is *time*. The consequences of an emission or impact may never end, even if our possibilities of following and modeling them decrease as time elapses after the intervention. It is particularly important to recognize the depreciation of future impacts achieved by narrowing system borders or (as economists do), by discounting, when dealing with global warming effects or depletion of natural resources (Azar and Sterner 1996). Yet another dimension is *space*. There are many examples of local environmental issues having been 'solved' by shifting the impact to another scale or a wider region.

If we choose to use global system borders, we must face the problem of trade-offs between local and global impacts. In impact evaluation, trade-off problems are ubiquitous, even if they are not always explicitly identified. For instance, when deciding to include an impact indicator in the study, there has to be some kind of weighting of its significance compared to other indicators or compared to some reference. In impact evaluation, as in many other types of evaluation, there are two ways of handling trade-offs. One is to try to minimize or maximize an objective function of some sort. This may be called a 'utilitarian' approach. (In modern politics it is often associated with the right wing.) Another is to try to achieve some type of justice, that is to deal with each indicator separately and try to reach an acceptable compromise. This notion is close to Herbert Simon's ideas of 'satisficing' in contrast to 'optimizing'. (In modern politics this worldview is mainly associated with the left wing.) In LCIA used for design purposes the utilitarian approach is often used (that is, the overall best option is sought) while in RA and EIA the latter approach is more often used. Of course, in practice, combinations of the two tradeoff types are common.

The way of handling uncertainty depends on the study context, but also on the practitioner's general attitudes. A common way is to let the degree of uncertainty decide whether an issue or figure should be included or not in the evaluation. Another, sometimes more fruitful, way would be to accept uncertainty as a part of reality and try to describe its consequences. Instead of focusing on what is 'correct', or not, one may ask what our present knowledge, in terms of data and models, tells us. The 'precautionary principle' is often used in impact evaluation and it works well with the 'justice' type of trade-off approach, but, for a utilitarian approach, safety margins in one impact type tend to decrease the appreciation of other impacts.

In IE we look for strategies to decrease the environmental impact from technical systems. Because normative aspects, such as choice of system borders, are of such an importance, these must be identified, handled in a systematic way and reported to the reader/decision maker.

SELECTION OF IMPACT INDICATORS

The selection of indicators reflects the goal and scope in terms of choice of system borders, the interpretation of the precautionary principle and the intended means of integration towards a total impact value. In LCIA impact indicators may be chosen anywhere along the cause–effect chain. For example, emissions and use of resources may be used directly as impact indicators and evaluated against what is normal or against national emission goals and so on. Or indicators may, as in CBA, be chosen late in the cause–effect chain to reflect those issues that are observable and known to ordinary people, such as excessive mortality or fish kill. In RA and EIA the selection of indicators is largely made according to praxis and not dealt with as an explicit procedural step, as in LCA.

In RA the indicators are mostly a ratio between two numbers. The numerator is an estimated concentration that may occur in a certain compartment (prognosticated environmental concentration, or PEC) of the environment. The denominator is an estimation of a 'no effect level' (prognosticated no effect concentration, or PNEC) or an 'acceptable' level arrived at by some informal process. A ratio greater than unity is an indicator of risk. (Many substances, such as carcinogens, may not have a finite 'no effect level', which implies that any measurable concentration indicates risk.)

In EIA the indicators are seldom emissions but may be concentrations in the environment or observable changes, like decline in tree growth and decreased biodiversity. The position of indicators along the cause–effect chain may vary in EIA between different impact issues and is often determined from what is available and practicable.

In the LCIA standard (ISO 14042) (ISO 2000) the selection of impact categories and category indicators is *required* to be consistent with the goal and scope, justified and reflect a comprehensive set of environmental issues related to the product system being studied. It is also required that the category indicator names be accurate and descriptive, that references be given and that the environmental mechanism linking the emission or resource use to the category indicator be described. It is further *recommended* that the indicators be internationally accepted, represent the aggregated emissions or resource use of the product system on the category endpoint(s), avoid double counting and be environmentally relevant. It is also recommended that value choices and assumptions made during the selection of impact categories and category indicators be minimized. The selection of category indicators at the same level in the cause–effect chains may help to avoid double counting.

MODELING OR RECOGNIZING INTERACTIONS BETWEEN TECHNICAL SYSTEM INDICATORS AND IMPACT INDICATORS

In RA this is the 'hazard identification' and 'risk assessment' step. In EIA and CBA the modeling or recognition of interactions is the core of the analysis. In LCA terminology, the corresponding steps are called 'assignment to impact categories' and 'characterization of impacts'.

Some models may be quantitative while others may be qualitative. The selection of models depends on goal and scope. In LCA the selection of models is subject to the same requirements and recommendations as the selection of impact categories and category indicators. Besides, the appropriateness of the characterization models used for deriving the category indicator in the context of the goal and scope of the study needs to be articulated.

The requirement of addressing the model performance is important. Sometimes one encounters the idea that models should be very similar to reality. Most models used in practice are, however, drastic simplifications of real processes. This may be evident to experts but not to laymen, who may constitute a large part of the audience. The issue, in practice, is not whether to simplify, but how? Thus, ideally, models should be accompanied by 'use manuals' indicating what assumptions have been made and for what sorts of problems these assumptions are (probably) legitimate and, conversely, for what sorts of problems they are not.

Models may be of several types. In EIA site-specific dispersion models are often combined with dose-response models to identify if and where negative impacts may occur. In RA similar types of dispersion models are used together with compartment models where the distribution between different compartments is specified. In LCIA, models in use are almost exclusively linear. Some models express potential effects and exclude fate. For instance, the 'acidification potential' of an emission is often defined as the maximum relative amount of hydrogen ions ($H+$ or protons) that may be released. In reality only a part of the $H+$ may be released, or the protons may be deposited in a limestone-rich area, where no acidification problems exist. LCIA characterization models may be of three types: (a) mechanistic (Figure 13.2), including dispersion and dose-response; (b) empirically based, including statistically significant correlations between a polluting substance and its effect; or (c) of an equivalency type, such as acidification potential.

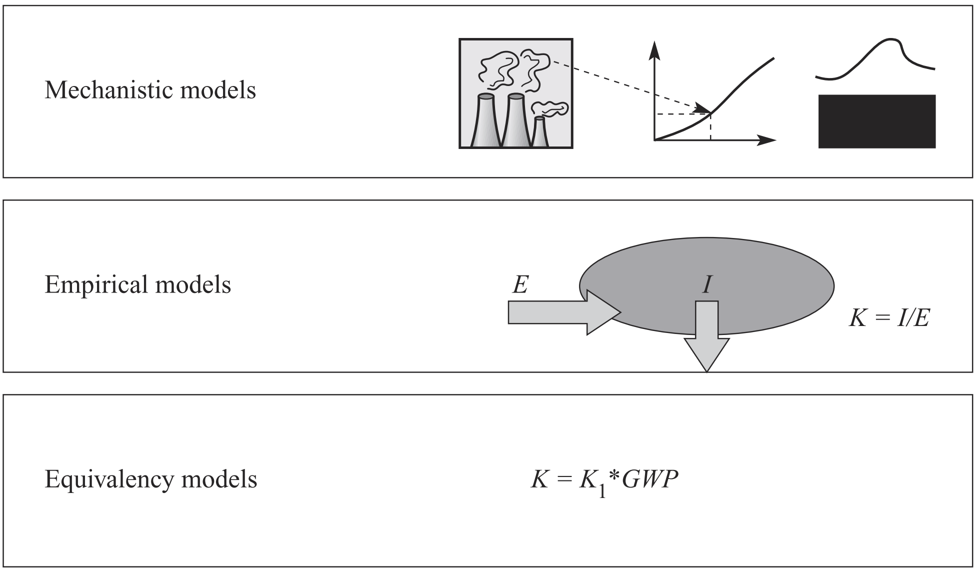


Figure 13.2 Different types of characterization models

Linear LCIA characterization models have been published for a number of substances (Lindfors *et al*. 1994; Hauschild and Wenzel 1998; Goedkoop and Spriensmaa 1999; Steen 1999). As the location and size of the sources are normally unknown, the characterization factor is in reality a distribution, which may be represented by an average outcome and a standard deviation (Figure 13.3). For a single emission, the uncertainty may be large, but when used for evaluating large technical systems the uncertainties tend to decrease, as a result of averaging.

COMPARING DIFFERENT TYPES OF IMPACTS AND EVALUATION OF TOTAL IMPACT

In LCA terminology evaluation can be made in several ways, such as normalizing, ranking, sorting or weighting. Earlier the term 'valuation' was used instead of weighting, but ISO found the term 'weighting' to better represent all techniques that were used. Normalizing is a standard procedure in RA, where an estimated concentration that may occur in the environment is always compared to a reference concentration. In a way ranking and sorting are also standard procedures in RA and EIA, where the overall aim is to elucidate significant environmental impacts.

Weighting is used systematically only in CBA and LCIA. In the standardization of

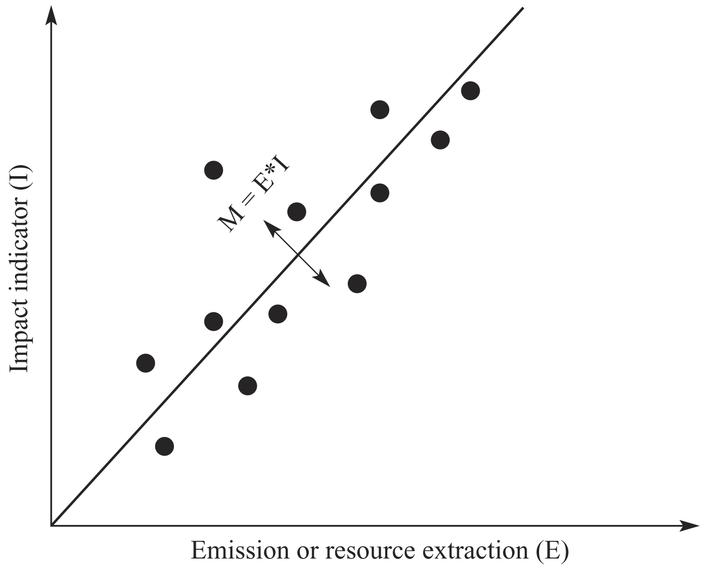


Figure 13.3 Relations between emissions and impacts may vary owing to location and other circumstances

LCIA, weighting has been a controversial step. ISO 14042 explicitly states that 'weighting shall not be used for comparative assertions disclosed to the public' (ISO 2000). 'Comparative assertions' are defined as 'claims of overall superiority'. In other words, the use of weight factors for evaluation is not sufficient to conclude that product A is superior to product B.

A cautious attitude is easy to understand if weighting results are seen as a 'verdicts' and if companies and responsible persons have limited ability to adapt to the 'law'. Representatives of the third world often mentioned LCIA and weighting in particular as a potential trade barrier. The industrial world might conceivably impose new requirements on the third world industry that it could not fulfill. However, if weighting is seen as comparing the overall outcome with different general environmental goals and public preferences, it may be less controversial (Bengtsson and Steen 2000). Those in favor of weighting claim that no choice between technical concepts can be made openly and transparently without weighting. If a formal weighting procedure is used the result is open for discussion and criticism. This is particularly valuable for a democratic process, as when the government develops guidelines for recycling or for use of some materials.

Economic evaluation, as in CBA, is common, but no less controversial. On one hand there is a wish to reach the vast number of decision makers who cannot understand and use environmental impact information unless it is expressed in economic terms. On the other hand, economic thinking has to a large extent put us where we are today in its inability to detect some of nature's core values. Is it possible to value what is priceless? A common criticism of economic valuation is based on the use of discounting. Discounting may significantly reduce the appearance of long-term effects such as global warming (see, for example, Azar and Sterner 1996)

Attitudes towards environmental changes vary considerably among people, sometimes with orders of magnitude. However, as with characterization factors, we are normally dealing with large technical systems influencing lots of people. Therefore the uncertainty decreases if we perform the analysis on the population level. A particular problem occurs when applying today's attitudes, which normally are available only from Western countries, to other cultures and future generations. As long as this is on a conscious level and reported in a transparent way, the problem may be handled, but there are many examples of technical projects that have not been sufficiently aware of local culture.

ANALYSIS OF UNCERTAINTY AND SENSITIVITY

Analysis of uncertainty and sensitivity is important but too seldom carried out. Life cycle assessments are normally made without quantitative estimations of accuracy or precision. In SETAC's 'Code of practice' (1993) sensitivity and uncertainty analysis are recommended, but the methodology is not very well developed. In the ISO 14040 sensitivity analysis is requested (ISO 1997a).

The topic has attracted attention recently. Hoffman *et al.* (1994) reviewed statistical analysis and uncertainties in relation to LCA discussing technical, methodological and epistemological uncertainty (for example, from lack of knowledge of system behavior). Heijungs (1997b) developed a sort of sensitivity analysis called 'dominance analysis', where the most important contributions to the result are identified. Kennedy *et al.* (1996) make an uncertainty analysis of the inventory part of an LCA using beta distributions. Steen (1997) developed a technique to estimate the sensitivity of ranking to uncertainty in input parameters.

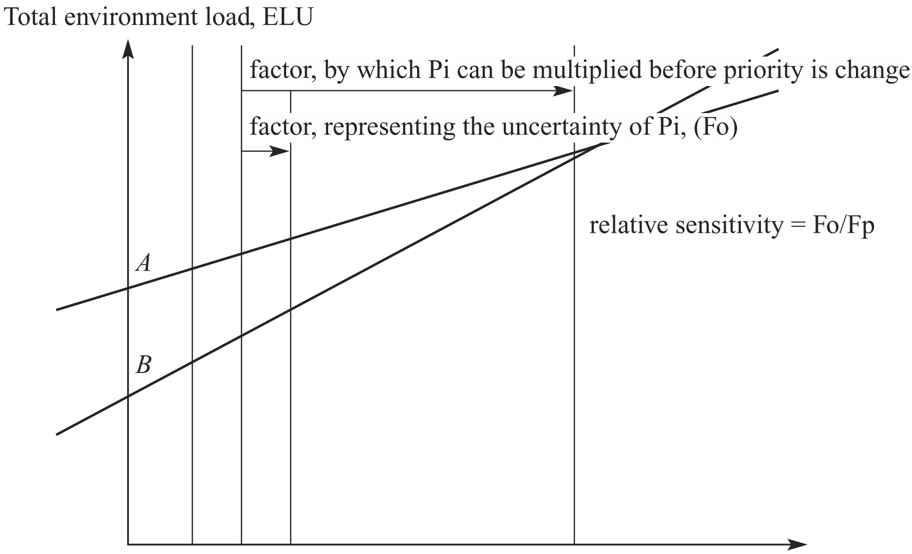
The outcome of an impact evaluation may be uncertain owing to uncertainties in input parameters, models or system borders. Depending on the goal and scope, more or less uncertain data and models may be included. Uncertainties in input data may be of several types: (a) uncertainties due to sampling from a population with true variation; (b) epistemological uncertainties (using data in another context than that where it was generated); and (c) measurement errors. It is often claimed that estimating uncertainty is itself too uncertain, and this is in turn used as an argument to omit an uncertainty analysis. It must be remembered, though, that the difference in uncertainty between different input parameters can be very large, as much as several orders of magnitude. It is of great value to find out if the result of an impact evaluation is accurate within a few per cent or whether it can vary by an order of magnitude or more.

Relative impact evaluations are less sensitive to input data uncertainties than absolute impact evaluations. If, for instance, there are linear relations between interventions from the technical system and impacts, all aggregated impact values will have the form

$$\Sigma\, i_j \,.\, k_{jk} \,.\, v_k$$

where i_j is the jth inventory result, k_{jk} the characterization factor between inventory parameter j and impact indicator k, and v_k is the weighting factor for impact indicator k.

Any change of a parameter i, k or v will thus result in a linear response of the aggregated result (Figure 13.4). If, for instance the parameter is a characterization factor and



Note: Pi is the value of parameter near i, Fo is the uncertainty of the established parameter value, Fp is the change in Pi that can occur before priority changes.

Figure 13.4 The aggregated impact value is linearly dependent on all input data

the corresponding inventory parameter value for alternative *A* is less than for alternative *B*, the corresponding slope is less. Clearly the priority will change if the characterization factor value increases beyond a certain level. The ranking will thus be less sensitive to uncertainties in impact evaluation factors *k* and *v*, which are normally used for evaluating both *A* and *B*, than for errors in *i*, which is presumed to be unique to either *A* or *B*.

Once there is an impact evaluation model established, the sensitivity of any of the outcomes to any of the inputs may be calculated. A common way of using sensitivity results is to sort them in order of size or magnitude. Then one may find the most important input parameters and this may yield ideas for improvement of technology or input data. However, some of these finding may be quite arbitrary, resulting from the way primary data were sorted and aggregated. For instance, suppose the sensitivity of an impact evaluation turned out to be much greater for emissions from the USA than from Luxembourg. If such a sensitivity analysis were used for directing abatement measures for the USA and not for Luxembourg, it is easy to see that something is wrong. Unfortunately, it is not always as easy to recognize such nonsense results when dealing with data from industrial systems.

DATA DOCUMENTATION AND REPORTING

Impact evaluation is complex and involves a great deal of data and written information. One of the major obstacles to the implementation and use of impact evaluations has been that they are difficult to survey and summarize. A key concept in the development of LCA has therefore been ‘transparency’.

In computer science there is a technique of structuring information by using data modeling, which is a formal technique with its own language. There is also a type of model that does not require knowledge in data model language. An example of this technique, as used by Carlson *et al.* (1998) and Carlson and Steen (1998), is shown in Figure 13.5. The basic elements in a data model are objects. Objects represent things you want to store information about. Objects are chosen to be unique and not overlapping. Between objects there are relations.

There are various database techniques to store information. A common type is the relational database. In that, information about the various objects can be stored in tables, and each object can have its own table. Redundancy is avoided through special references, or 'keys', that are used in all other tables except the one where the original information is stored. In ISO's work on the LCA standards, there is, at present (c.2000), a special working group developing a common data documentation format which is intended to appear as the standard ISO 14048. The standard is at present focusing on the description of technical systems and their emissions and resource flows, but the technique may also be extended to impact assessment.

A central object in Figure 13.5 is the *category indicator*. It belongs to one or several *impact categories*, which in turn may belong to each other in a hierarchy. Indicators are chosen according to an *impact indication principle*. In the modeling of relations between emissions and impacts, *characterization parameters* are used. They may be of different *characterization parameter types*, one of which is the characterization factor. Each characterization factor is determined by a special method. Information about this is found in the table, *characterization method*. Information about *weighting factors* is stored in a special table with references to the *weighting method* (or principle) it belongs to.

An impact assessment or evaluation of the impact is made by selecting and combining indicators, characterization factors and weighting factors for a given type of emission or resource flow. This is reported in the table, *flow group impact assessment*, where references to indicators and so on are made together with a reference to the *flow group* at hand. For practical reasons it is an advantage of being able to store information about 'ready-made' impact assessments for certain types of emission and so on. These types are identified in the flow group table, with respect to important parameters governing their impact, such as location and stack height. Each flow group impact assessment belongs to an *impact assessment method*. In each impact assessment of a specific technical system, an *activity impact assessment*, is reported which method was used for the specified activity. Information about the activity is stored in the life cycle inventory database, as is general information about flows. The rest of the database structure, named SPINE, is reported by Steen *et al.* (1995).

Although systematic and harmonized data documentation formats are only occasionally used in impact assessment today, it must be seen as one of the most interesting developments in the near future and a key to successful data exchange and use of software.

DISCUSSION

Many of the impact assessment methods in use are still relatively young and immature. Looking at the science of environmental impact evaluation in terms of Hegel's model of

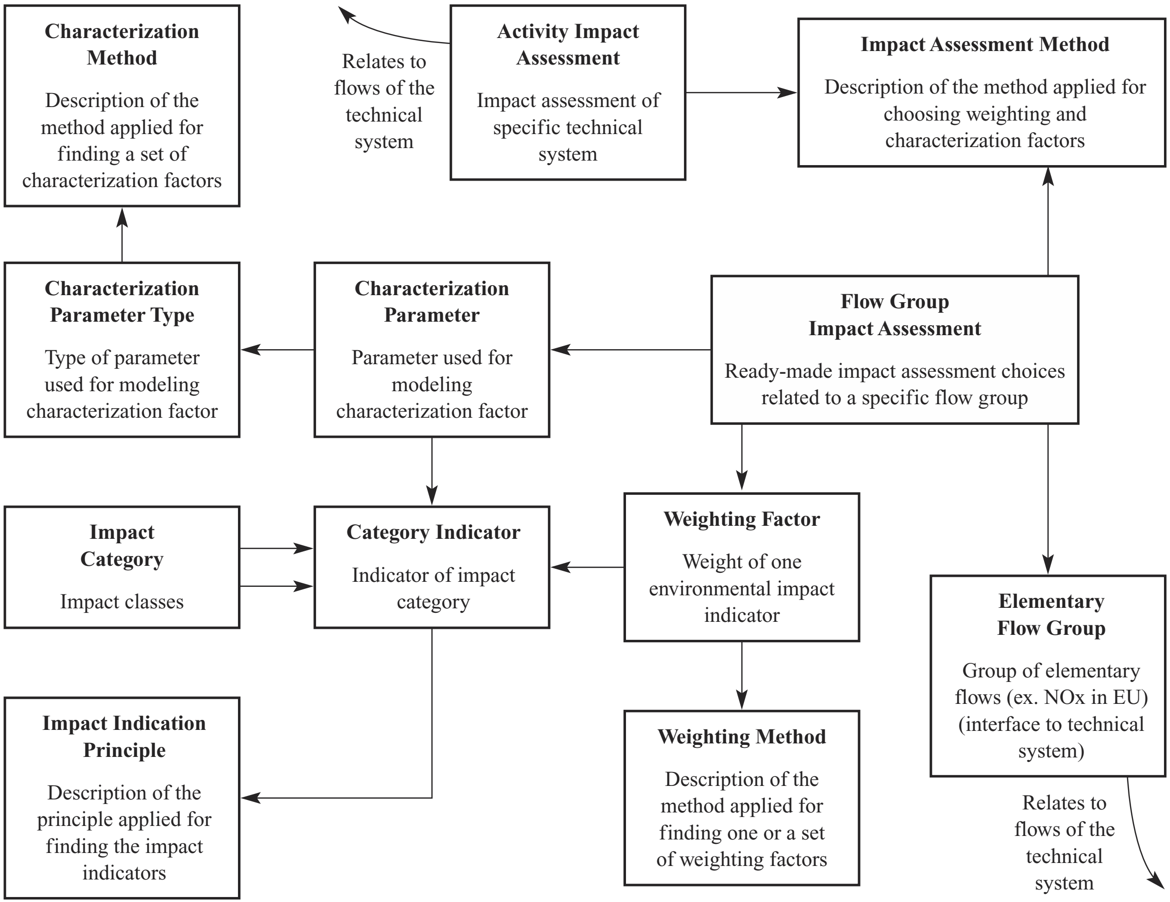


Figure 13.5 Conceptual data model of impact evaluation

successive phases of complication and integration, it appears that the complication phase has, so far, been dominant. The evidence for this is the ever-increasing number of methods and parameters being introduced as well as from the increased number of methods suggested for weighting of different impact types. However, the integration phase is now beginning to develop. This may be seen from an increasing demand for a simpler and a more comprehensive language.

When describing impact evaluation there is a risk of making it into 'a science of acronyms'. It seems that a common way of addressing a complex problem like impact evaluation is to agree about a procedure instead of an analysis. Procedures may be understood by many. Understanding an impact analysis is much more a qualified expert task. The preference for agreements about procedures may be seen in the ISO 14000 project, which is entirely about environmental management. The LCA standards are purely procedural.

It must be remembered that having a good procedure is only helpful up to a certain level. The quality assurance described in the ISO 14000 series is an assurance only of the procedures, not of the result. The basic problems of finding relevant data for interventions, identifying and modeling cause–effect chains and describing human attitudes to impacts still have to be solved.

Normally the impact assessment is made from 'left to right', that is starting with interventions you know and trying to identify and assess their consequences. Most of the interventions that have been registered have been registered because they 'may' have some impacts on the environment. But there is an over-representation of measurements on interventions that have little actual impact on the environment, as they are part of monitoring programs aimed at preventing impacts. It is therefore also useful to go from 'right to left', that is to check all significant environmental issues and whether they may be influenced by the activity evaluated.

SUMMARY AND CONCLUSIONS

Several impact evaluation techniques have been developed for different situations. *Risk assessment* is used to decide upon rules and regulations about handling of chemicals. *Environmental impact assessment* is used to decide about rules and regulations for industrial plants, roads and other technical projects. *Cost–benefit analysis* is used to evaluate the meaningfulness of measures to decrease environmental impacts. *Life cycle impact assessment* is used to evaluate impacts from products or product systems.

A central concept in all evaluation techniques is the impact indicator. Different impact evaluation techniques have different names and pay different attention to the way the selection of indicators influences the outcome of the evaluation. Modeling or recognizing interactions between technical system indicators and impact indicators is a core element in an impact evaluation, but is linked to uncertainties that need to be recognized more explicitly.

Comparing different types of impacts and evaluation of total impact is made systematically only in cost–benefit analysis and life cycle impact assessment. Sometimes the results are very controversial as they are presented, or interpreted, as a 'final judgment'. This is inappropriate. These tools are merely one among several impact evaluation techniques.

The relatively high degree of complexity of the subject, combined with a moderate awareness of impact evaluation methodologies among non-experts, calls for a harmonization of language and methods, but without restricting the freedom of the analyst to choose the most appropriate methodology.

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PART III

Economics and Industrial Ecology

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14. Environmental accounting and material flow analysis

Peter Bartelmus

ASSESSING SUSTAINABILITY: A PROLIFERATION OF APPROACHES

Global warming and depletion of the ozone layer, land degradation by agriculture, industrial and household pollution, depletion of subsoil resources by mining, loss of habitat and biodiversity from deforestation, and desertification from grazing semi-arid lands are conspicuous examples of the impacts of economic activity on the environment. They are generally viewed as symptoms of the unsustainability of economic production and consumption, and many indicators have been advanced to confirm this. Table 14.1 shows some indicators taken from a large variety of international sources. They differ widely in concepts and definitions, scope and coverage, units of measurement, statistical validity and results. There is an obvious need to develop a common conceptual framework as a basis for more systematic data collection and analysis.

Table 14.1 Indicators of non-sustainability

Indicator	Estimate
Biomass appropriation of terrestrial ecosystems	40%
Climate change	1–3.5°C of global warming (2100)
	65cm sea level rise (2100)
Ozone layer depletion	30–40% decrease of ozone column above Antarctica
Land degradation	11% of vegetated surface degraded (since 1945)
	10 million environmental refugees
	500 billion tons of topsoil lost (since 1972)
Desertification	5 million ha of cropland lost annually
	70% of agricultural dryland lost
Biodiversity	¼ of total biodiversity in danger of extinction
	5000 to 150 000 species lost annually
Deforestation	16.8 million ha of forest area lost annually
Fossil fuels	90 years of proved recoverable reserves
	243 years of proved reserves in place
	800 years of total resources

Source: Bartelmus (1994, Table 1.3).

At first sight the common underlying notion of sustainable development seems to provide such a framework. Unfortunately, popular definitions such as the Brundtland Commission's 'satisfaction of current and future generations' needs' (WCED 1987) or the economists' favorite of 'non-declining welfare' (Pezzey 1989) are opaque: both fail to specify the ingredients and time frame of welfare or needs. Nor do they specify any particular role for the environment. No wonder scarcely comparable indicators of the quality of life (Henderson, Lickerman and Flynn 2000), sustainable development (United Nations 1996a), human development (UNDP 1999), genuine progress (Cobb, Halstead and Rowe 1995), expanded wealth (World Bank 1997), ecological footprints (Wackernagel and Rees 1996) or environmental sustainability (Yale University *et al.* 2000) have proliferated.

A further obstacle to agreeing on common indicator sets and a common strategy for sustainable development is a prevailing polarization of environmental and economic scientists who seek to impose their own particular values on the counterpart field. This mutual colonization also seems to continue within the overall rubric of economics, as resource, environmental and ecological economists apply their own cherished tool kits to extend the boundaries of neoclassical economic analysis (Bartelmus 2000). Environmental economists attempt to put a *monetary* value on the loss or impairment of environmental services as a first step towards 'internalizing' these 'externalities' into the budgets of households and enterprises. Green accounting systems are among the more systematic attempts at modifying conventional macroeconomic indicators such as GDP or capital formation. Most environmentalists and even some ecological economists, on the other hand, reject the 'commodification' and pricing of the environment. In their view, the value of the environment cannot be expressed in money. For them, *physical* indicators of sustainable development, such as those of Ayres (1993b, 1996); Azar, Holmberg and Lindgren (1996); Ayres and Martinàs (1995); United Nations (1996a); or Spangenberg *et al.* (1999) are preferable.

Physical indicator lists do cover a broader set of social values and amenities. They do not have, however, the integrative power of monetary aggregates generated in environmental accounting systems. But policy makers prefer highly aggregated indices to get a picture of the forest rather than looking at particular trees. When monetary valuation is disdained, more compound indices are constructed, usually as indicator averages, as for instance by UNDP's (1999) Human Development Index, or by adding up the weight of materials entering the economy, notably in material flow accounts (described below). This chapter discusses some of the pros and cons of two commonly applied physical and monetary approaches, with a view to linking or combining them.

PHYSICAL AND MONETARY ACCOUNTING: COMMONALITIES AND DIFFERENCES

Concepts and Methods

Among the above-mentioned indicator frameworks and index calculations, two systemic approaches appear to have become widely accepted standards for assessing the environmental sustainability of growth and development. They are the physical material flow accounts (MFA), developed for particular commodities by the US Bureau of

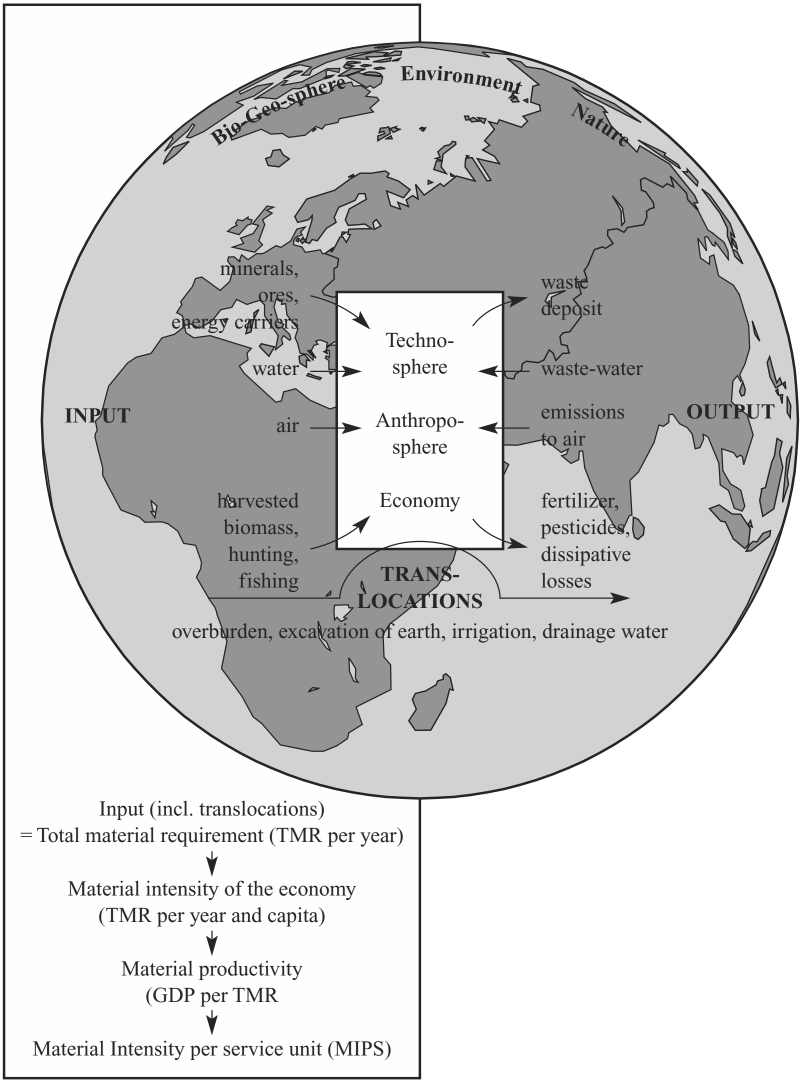
Mines over a period of decades (USBM 1970, 1975, 1985) and generalized to the national level by the Wuppertal Institute for Climate, Environment and Energy (Bringezu 1997a, 1997b; Schmidt-Bleek *et al.* 1998; Spangenberg *et al.* 1999) and the physical and monetary System of Integrated Environmental and Economic Accounting (SEEA) of the United Nations (1993a). For a summary description, see Bartelmus (1999). The SEEA is designed as a 'satellite' system of the worldwide adopted System of National Accounts (SNA) (United Nations *et al.* 1993) with which it maintains greatest possible compatibility. Such compatibility with a standard accounting system has not yet been achieved for the MFA. It is addressed in the revision of the SEEA by the so-called 'London Group' of national accountants through link-up with physical accounting approaches. For the present status of the revision process, see the home page of the London Group, http:www.statcan.ca/citygrp/london/publicrev/intro.htm

Figure 14.1 illustrates in a simplified manner the approach to material flow accounting. Material throughput through the economy is shown as inputs of material flows from abroad and the domestic environment, and outputs of residuals discharged into the environment and of materials exported to the rest of the world. This balance of inputs into, accumulation of materials in, and outputs from the economy includes also so-called 'translocations' or 'ecological rucksacks' which are indirect flows that do not become part of a product but which are concomitant to its production (Spangenberg *et al.* 1999, pp. 15–16). The MFA assess the use and movement of materials by means of one key indicator, the total material requirement (TMR) and several derived indicators, notably the material intensity (MI) of the economy, measured as TMR per capita and per year, material intensity per unit of service (MIPS) and the material productivity of the economy, GDP/TMR. The MIPS analysis was developed by Schmidt-Bleek (1992a, 1994a). An overview is given by Liedtke *et al.* (1998).

The SEEA, on the other hand, attempts to incorporate the key functions of natural capital, that is resource supply, waste absorption and use of space, into the asset and production accounts of the national accounts. Figure 14.2 shows how the SEEA is derived from the standard national accounts as an expansion of conventional stock (asset) and flow (supply and use) accounts. Environmental components are added by incorporating environmental assets and asset changes in the shaded vertical column of the asset accounts. At the same time, natural resource depletion and environmental quality degradation represent additional environmental costs in the use accounts, as indicated in the shaded row of natural asset use. Environmental costs reflect the consumption of natural capital and are therefore recorded in both the asset and flow accounts. In this manner important accounting identities, and hence the system character of the accounts, are maintained. Finally, expenditures for environmental protection are shown as 'thereof' elements of conventional aggregates (see Figure 14.2; they represent a social response to environmental impacts.

The inclusion of natural assets and asset changes in national accounts generates environmentally modified monetary indicators. Summing up the rows and columns of Figure 14.2 yields most of these indicators. They include, in particular:

1. environmentally adjusted value added (EVA), generated by industries and calculated by deducting environmental (depletion and degradation) cost incurred by industries from their (net) value added;

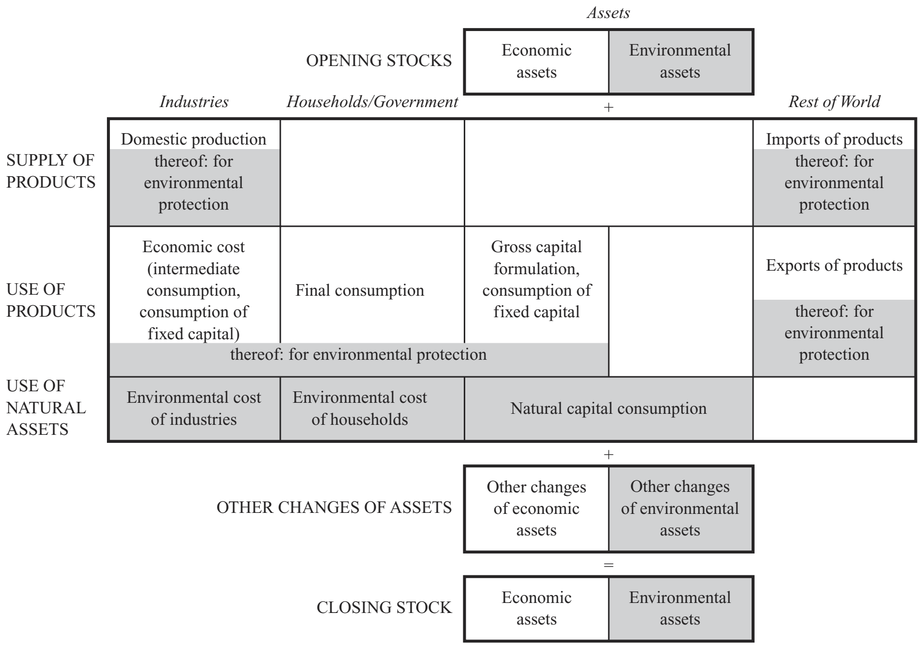


Source: Wuppertal Institute (after Bringezu 1993) UM-194e-2/93.

Figure 14.1 Material flow accounting (MFA)

2. environmentally adjusted net capital formation (ECF), obtained by deducting environmental cost from conventional (net) capital formation; and
3. environmentally adjusted net domestic product (EDP), obtained by deducting environmental cost from net domestic product (NDP) or calculated as the sum of final consumption, ECF and the balance of exports and imports.

Note that these indicators comply with the accounting identities of the conventional national accounts. EDP can thus be calculated as the sum of final demand categories



Source: Bartelmus (1999, Figure 2).

Figure 14.2 SEEA: flow and stock accounts with environmental assets

(capital formation, final consumption and net export) or of value added generated by industries. These and other identities provide a valuable check on the consistency of concepts and definitions, and the validity of the data collected. Such checks are, of course, missing in physical indicator frameworks such as those of the OECD (1994a) or the United Nations (1996a), as well as for index calculations outside the national accounts. For example, the Human Development Index is an average of one monetary indicator (GDP per capita) and two non-monetary indicators of life expectancy and literacy (UNDP 1999). The selection of indicators and inherent equal weighting of unequal issues impairs the validity of such indices, including the above-mentioned genuine progress and environmental sustainability indicators.

The Valuation Controversy: Pricing or Weighing?

Putting a monetary value on natural assets and their changes, even if they are not traded in markets, is a prerequisite for establishing the above-mentioned accounting identities and calculating their component indicators. However, the imputation of monetary values for environmental phenomena, which were not necessarily observed in markets, has been criticized, not only by environmentalists, but also by more conservative national accountants. The following paragraphs review briefly, therefore, the three commonly proposed valuation techniques as to their capability of assessing environmental impacts and repercussions.

Market valuation

As the name suggests, market valuation uses prices for natural assets which are observed in the market. It is usually applied to 'economic' assets[1] of natural resources, though trading of pollution permits could also generate a market value for 'environmental' assets of waste absorption capacities. Where market prices for natural resource *stocks*, such as fish in the ocean or timber in tropical forests, are not available, the economic value of these assets can be derived from the (discounted) sum of net returns, obtained from their potential use in production. This is the value at which a natural asset. such as a mineral deposit or a timber tract, would be traded if a competitive free market existed for the asset. Market valuation techniques are also applied to *changes* in asset values, caused in particular by depletion, that is non-sustainable asset use. These value changes represent losses in the income-generating capacity of an economic asset. Depletion cost allowances thus reflect a *weak sustainability* concept, calling for the reinvestment of imputed environmental costs in any income-generating activity of capital formation or financial investment.

Maintenance valuation

Maintenance valuation permits the costing of losses of environmental functions that are typically not traded in markets. Dealing only with marketed natural resources would drastically limit the scope of economic analysis concerned with scarce goods and services, whether traded or not. In industrialized countries, especially, environmental externalities of pollution can be of far greater importance than natural resource depletion. The SEEA defines maintenance costs as those that 'would have been incurred if the environment had been used in such a way as not to have affected its future use' (United Nations 1993a, para. 50).

Maintenance costs are the 'missed opportunity' costs of avoiding the environmental impacts caused during the accounting period. They refer to 'best available' technologies or production processes with which to avoid, mitigate or reduce environmental impacts. Of course, these costs are hypothetical since environmental impacts did occur. They are used, however, to determine weights for actual environmental impacts generated during the accounting period by different economic agents. Those agents did not internalize these costs into their budgets but *should have done so* from the societal point of view. As with depreciation allowances for the wear and tear of produced capital, such costing can be seen as a way of identifying the funds required for reinvesting in capital maintenance.

Actual internalization would of course change consumption and production patterns. The ultimate effects of internalization could be modeled in order to determine hypothetical aggregates such as 'analytical green GDP' (Vu and van Tongeren 1995) or an 'optimal net domestic product with regard to environmental targets' (Meyer and Ewerhart 1998a).

Damage valuations

Damage valuations and related, notably contingent valuations were also proposed in the SEEA for environmental accounting. They were applied in cost–benefit analyses of particular projects and programs but are hardly applicable in practice at the national level. They refer to ultimate welfare effects (that is, damages) of environmental impacts that are inconsistent with the pricing and costing of the national accounts and quite impossible to trace back to causal agents. Contingent valuations which express a willingness to pay for damage avoidance are inconsistent with market prices because of their inclusion of consumer surplus. They also face well-known problems of free-rider attitudes and consumer ignorance. Mixing these 'cost-borne' valuations with 'cost-caused' (maintenance cost) valuations creates aggregates which are neither performance nor welfare measures and therefore difficult to interpret.

Conservative national accountants and economists, especially those in industrialized countries, have been quite recalcitrant in implementing environmental satellite accounts in monetary terms. While some now favor the incorporation of the cost of natural resource depletion into the conventional accounts (Hill and Harrison 1995), many consider the costing of environmental externalities a matter of 'modeling' which, with few exceptions, is deemed to be off limits for 'official' statisticians (van Dieren 1995; Vanoli 1998). The reason is that national statistical offices believe they might lose some of their long-standing 'goodwill' from clients (such as finance ministries) if they introduced controversial concepts and valuations, even through supplementary satellite systems.

As a result, a number of relatively timid approaches of mixed (physical and monetary) accounting have now been adopted, mostly in Europe. The prototype is the Dutch National Accounting Matrix including Environmental Accounts (NAMEA) (Keuning and de Haan 1998). It refrains from monetary valuation of environmental impacts by simply allocating physical measures of environmental impacts (mainly emissions) to responsible economic sectors. This approach facilitates the linkage of physical impacts with their immediate causes. It fails, however, in aggregating these impacts and relating them as capital consumption and accumulation to the balance sheets of natural assets. To improve on this situation, that is to enhance the policy relevance of the physical data, the NAMEA authors combine different environmental impacts by means of 'environmental policy theme equivalents'. However, these aggregates suffer from limitations in selecting and defining the themes, and their equivalent factors which still do not permit inter-theme comparison.

The above-described MFA attempt to resolve the aggregation problem for physical measures by assessing material flows with their 'natural' (mass) unit of measurement: weight. Such weighting by weight has been criticized as 'ton ideology' since counting tons reduces all kinds of environmental hazards caused by one factor, material input, to a simple one-dimensional measure of this factor. It can be argued that difficult-to-predict potential environmental impacts are best addressed by an indicator like TMR, which focuses on the origin of these impacts, extraction and use of materials, in a highly visible fashion.

For a comprehensive critique of MFA, see Gawel (1998) and, for a counter-critique, Hinterberger, Luks and Stewen (1999). It must be acknowledged that mass is not the only way of measuring materials flows. Weight does not reflect the amount of energy flows, even if energy carriers are included in material flow categories. An alternative, with strong theoretical support, is the thermodynamic concept of exergy, first proposed as a measure of resource flows by Wall (1977, 1987, 1990) and extended by Ayres *et al.* (1998).

Physical and Monetary Aspects of Sustainability: Dematerialization and Capital Maintenance

Consistent with their focus on physical and monetary data, MFA and SEEA also reflect different notions of the sustainability paradigm, which may be more difficult to reconcile.

They can be categorized as the needs for *dematerialization* of economic activity and for the *preservation of natural and produced (fixed) capital* assets, used in production.

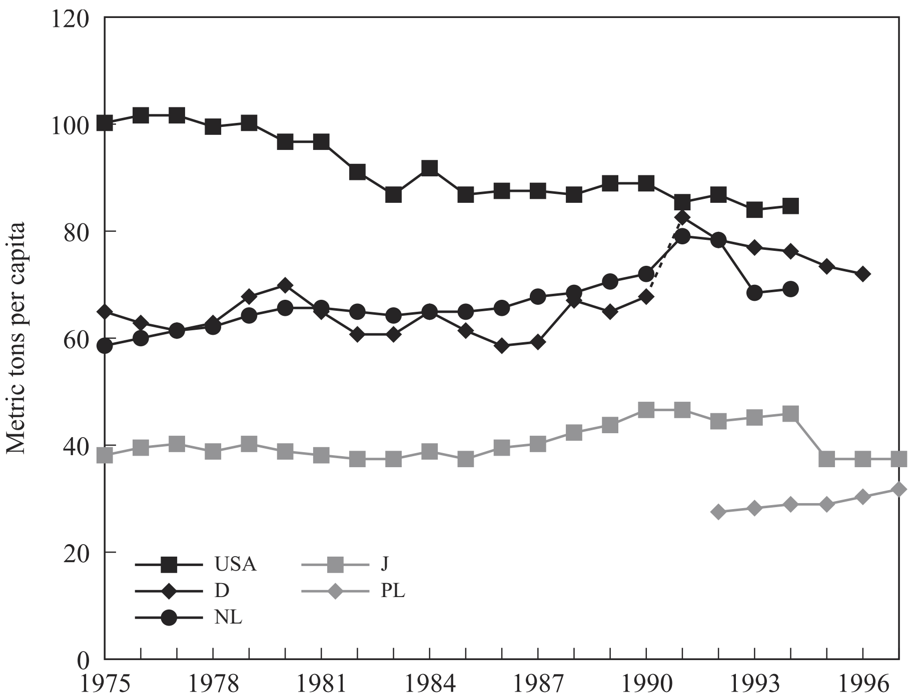
Both the TMR and MIPS indicators of material flow accounting (MFA) reflect the total use of materials as an index of throughput through the economy, including their hidden 'ecological rucksacks'. For achieving sustainability of economic performance such throughput should be at a level compatible with the long-term 'ecological equilibrium' of the planet. Ecological equilibrium can be operationalized by applying the normative notion of equal 'environmental space', that is, access to environmental services by everybody, to the overall dematerialization effort. One result is a sustainability standard calling for halving global TMR while doubling global wealth and welfare: the popular notion of *Factor 4* (Weizsäcker *et al.*, 1997). Under current production and consumption patterns, this can be translated into a *Factor 10* for industrialized countries. The assumption is that an equal environmental space should be reached by all countries in about 50 years while permitting a limited increase of material use in developing countries (Schmidt-Bleek 1994a, p. 168). It is recognized that such norms, which are based on reducing the total weight of materials used, are 'unspecific' in their attempt at reducing overall environmental pressure. On the other hand, all kinds of actual and potential environmental impacts and welfare effects are captured, at least roughly. In this manner, a precautionary approach is applied which permits anticipating potentially disastrous and largely unknown environmental effects (Hinterberger, Luks and Stewen 1996).

By contrast, economic accounting does not deal with uncertainty. It is a statistical information system which measures economic performance during a past accounting period. With regard to physical depletion and degradation of natural assets, the SEEA measures only actually occurred and specific impacts of natural resource losses and pollution, generated by different economic activities. The setting of normative standards is thus avoided in principle, since the deduction of the value of natural capital consumption can be seen as compiling only a 'net' value of production, without double counting of (depreciation) costs. Even though capital loss was not avoided de facto, the generation of (hypothetical) funds by means of a depreciation allowance would permit reinvestment of these funds for new capital formation. Such accounting for capital maintenance extends the sustainability criterion – allowing for capital consumption – already built into the conventional indicators of national income, product and capital formation, to natural capital. As shown above, modified aggregates of EDP, EVA, ECF, environmental cost and wealth (in economic and environmental assets) are the result of such accounting.

RESULTS AND POLICY ANALYSIS

Dematerialization: Delinking TMR and Economic Growth

Reducing material flows in terms of TMR aims at decoupling economic growth from the generation of environmental impacts. TMR per capita seems indeed to be leveling off, for selected industrialized countries, at 75 to 85 tons per annum, except for Japan at 45 tons because of its low per capita energy use and lower erosion losses (see Figure 14.3).[2] Given that GDP per capita is increasing in all countries there is some delinkage, albeit far from the prescriptions of Factors 4 and 10. The tentative conclusion is that current delinkage



Source: Adriaanse *et al.* (1997) and updating by S. Bringezu and H. Schütz (Wuppertal Institute for Climate, Environment & Energy).

Figure 14.3 Annual TMR per capita for the USA, the Netherlands, Germany, Japan and Poland

cannot be equated with sustainability as specified by these physical/ecological sustainability standards.

Advocates of the so-called environmental Kuznets curve (EKC) hypothesis suggested that delinkage will be an 'automatic' feature of growth. The implication is that no further action is required, once a certain level of economic development is reached. Unfortunately, empirical studies confirm the EKC hypothesis only in selected cases and for particular emissions (Perrings 1998). (See also a special edition of *Ecological Economics* 1998.) It is therefore useful to recast dematerialization in more strategic terms for purposes of policy analysis.

One such term is *resource productivity* which focuses on new technologies to reduce material inputs while generating the same or even better ultimate services from outputs. Such an increase in resource productivity is the mirror image of a decrease in material intensity as proclaimed by the MIPS indicator. It is generally held, however, that technology alone cannot be the savior from non-sustainability: it needs to be reinforced by more or less voluntary restriction in consumption levels. 'Eco-efficiency' in production needs to be combined with 'sufficiency' in final consumption. Otherwise, efficiency gains could be offset by increased consumption, due to lower prices made possible by the very same efficiency gains.

The task of MFA and its indicators would be to monitor progress in eco-efficiency and sufficiency and supply the information needed to link such progress to policy instruments of dematerialization. Such monitoring seems to increase in relevance, the lower the level of analysis. Physical indicators are most useful at the (managerial) micro level. Here, particular materials can be easily linked to different production and consumption processes, and their potential impacts become more obvious. Eco-efficiency and MIPS are thus on target when considering production techniques in enterprises and consumption patterns of households. Moving up towards meso and macro levels of aggregation, the non-specificity of material flow aggregates makes it more difficult to base policy decisions on (the weight of) material flows. As a consequence, policy advice is deliberately couched in 'guardrails', suggesting a less stringent guidance in the use of materials towards Factors 4 or 10 (Spangenberg *et al.* 1999, ch. 7.2).

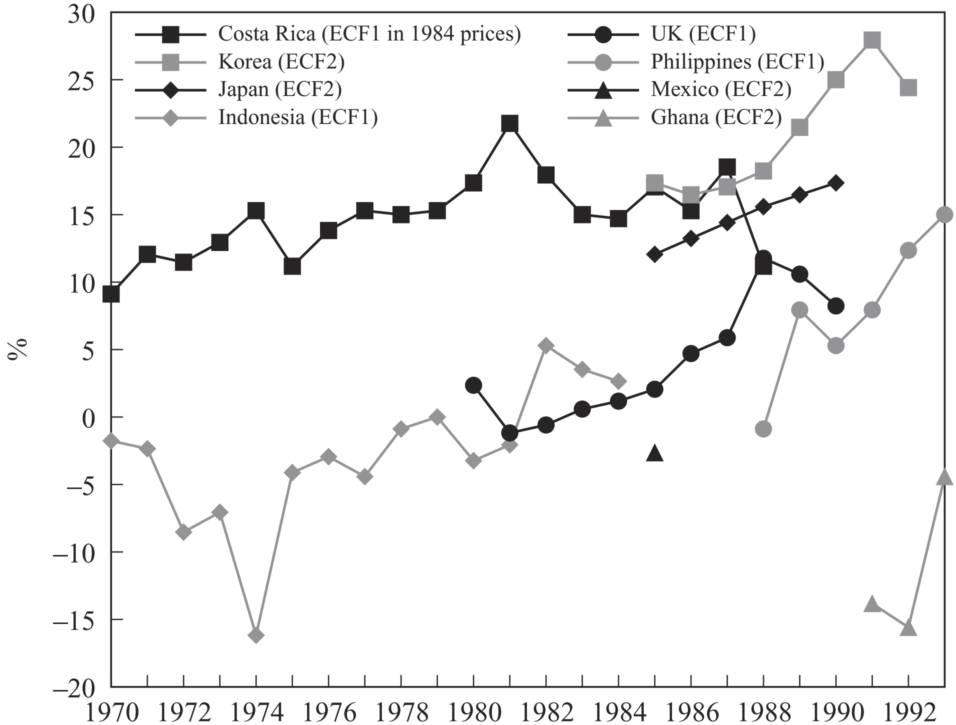
Proposed instruments range from voluntary restraint in the use of materials to eco-labeling of resource saving production processes and products. It is interesting to note that the instruments also include monetary (fiscal) (dis)incentives and hence a notion of cost internalization, focusing however on discouraging the use of physical inputs rather than on minimizing environmental impacts (Spangenberg *et al.* 1999, ch. 7.3).

Capital **Maintenance: Accounting for Accountability**

Costing natural capital consumption and thus allowing for the possible reinvestment of these costs reflects a monetary/economic notion of sustainability, namely as overall capital maintenance. Non-declining EDP would therefore indicate a (more) sustainable trend of economic growth. Compilations of EDP in case studies of environmental accounting (some of which are presented in Uno and Bartelmus 1998) did not show, at least for the countries included, a reversal in growth trends, comparatively measured by time series of GDP and EDP. One reason might be the relatively short time series available. Given this data restriction, a more pertinent way of looking into the sustainability of economic performance is to measure a nation's ability to generate new capital after taking produced and natural capital consumption into account.

Figure 14.4 presents environmentally adjusted net capital formation (ECF) in per cent of net domestic product (NDP). Indonesia, Ghana and Mexico (as far as a one-year result can tell) exhibited a non-sustainable pattern of disinvestment. The recent performance of all other countries seems to have been sustainable, at least for the periods covered, and in terms of produced and natural capital maintenance. This applies also to Germany, where the author recently estimated ECF/NDP to be positive and in the range of 8 and 10 per cent during 1990 and 1995, with environmental costs of about DM 60 billion or 3 per cent of NDP (Bartelmus with Vesper 2000)

Of course, such costing refers to the accounting and economic sustainability principles of keeping capital intact and does not represent welfare effects of, or damages to, the environment. Furthermore, past overall capital maintenance (or increase) tends to hide the fact that in the long run complementarities of natural capital might make it impossible to maintain current production and consumption patterns and growth rates. Extending past trends into the future thus reflects a 'weak sustainability' concept: the assumption is that natural capital can be replaced, at least 'at the margin'[3] by other production factors. The empirical testing of this assumption should be an important field of sustainability research.



Note: ECF1 covers natural resource depletion only; ECF2 covers depletion and degradation costs.

Source: Bartelmus (1997).

Figure 14.4 Environmentally adjusted net capital formation (ECF) in per cent of NDP

To encourage a strategy of capital maintenance at the *micro level* of enterprises and households the environmental costs of depletion and degradation need, first of all, to be allocated to those who generate the costs. For the second step, prompting economic agents into 'internalizing' these costs, most (neoclassical) economists favor market instruments such as fiscal (dis)incentives, or tradable pollution permits, over direct regulation. Theoretically, internalized degradation costs should reflect the ultimate welfare losses generated by environmental damage (to health and well-being), that is the costs *borne* by individuals. As discussed above, such damage costing is not practicable in (national) environmental accounting. Instead, maintenance costing is applied which assesses the cost of hypothetically avoiding actual impacts on the environment. Such costing permits us to allocate the macroeconomic social (expenditure) costs, generated by the degradation of a public good, to those who *caused* the degradation. In other words, polluters can be made 'accountable' for their environmental impacts, in line with the popular 'polluter pays' principle.

Environmental maintenance costs are thus those at which the market instruments should be set, initially and pragmatically. They refer to the best available 'eco-efficient' solution which could have prevented environmental impacts or reduced them to acceptable environmental standards. The ultimate effects of possible cost internalization on the economy, that is, their final incidence on other market partners and corresponding changes in production and consumption patterns, would have to be modeled in terms of assumptions about price elasticities and production and consumption functions.

At the *macroeconomic level*, the comparison of the availability of different categories of produced and non-produced natural capital facilitates the setting of priorities for increase, exploitation or maintenance of natural and produced wealth. The availability of productive wealth thus determines the long-term growth potential of an economy. A declining (natural) capital base would alert us to limits of growth, nationally and globally. The World Bank even considers comprehensive wealth assessments as a new model for 'development as portfolio management' (World Bank 1997, p. 28).

Changes in stocks through exploitation, discovery, growth, natural disasters and capital consumption are particularly important for investment decisions, as is capital productivity which includes natural capital. Capital productivity may change and differ (among different economic sectors) considerably after incorporation of natural resource stocks. Altogether different investment, price and growth policies should be the consequence of this information.

In addition, assessing the ownership of these stocks allows us to make informed decisions about establishing property rights for common-access resources. Such allocation might bring about a more caring treatment of environmental assets by its current users. More importantly, information about ownership of environmental assets would help arguing for a more equitable distribution of these assets among individuals, countries and the present and future generations. Striving for equity in this regard would reflect a new form of societal accountability in the management of environmental assets at local, national, global and intertemporal levels.

NOTES

1. In the sense of the SNA which defines 'economic assets' as 'entities (a) over which ownership rights are enforced . . . and (b) from which economic benefits may be derived by their owners' (United Nations *et al.* 1993, para. 10.2). Therefore Figure 14.2 displays part of natural capital consumption under the column of economic assets.
2. Note that Germany's reunification in 1990 increased material use abruptly. Since then, through adaptation to Western production and consumption patterns, material inputs decreased considerably in the 'new States'.
3. Pointed out by David Pearce at the Second OECD Expert Workshop on 'Frameworks to Measure Sustainable Development' (Paris, 2–3 September 1999), meaning that substitution of *total* stock is, at least in the short and medium run, not necessary as is sometimes assumed by critics of the weak sustainability criterion.

15. Materials flow analysis and economic modeling

Karin Ibenholt

A standard MFA gives an overview of the current, or even historical, material status in a country (or economy). But in order to approach issues like sustainable development, there is also a need to analyze possible future developments of material flows. This is especially true when analyzing how different policies (environmental and others) may affect the material flows in a society.

The flows of materials are to a large degree determined by the broad interplay between different agents (the consumers and producers) that characterizes economies today. There is, for instance, a large volume of deliveries inside and between the different production sectors. Changes in the end consumption of a product will have repercussions through most sectors in the economy, since it is not only the producer of the product that must change the production but also producers of intermediate goods and raw materials. When studying the use of materials in an economy it is important to consider this complexity. Economic models do attempt to handle this interaction between economic agents and can therefore be considered as suitable tools for predicting and analyzing the consumption of materials.

For the purpose of this chapter economic activity is considered to be a driver of material consumption, and not vice versa. In the real world, the causality is more likely to be two-directional.

When doing a forecast of material flows (subject to the above caveat) one has to choose a model that describes the society, or economy, that the forecast will cover. This model may be rather simple. For instance, it may just extrapolate existing, and historical, trends for the variables one is analyzing. An illustration of this method is to use an input–output model (see Chapter 10) and extend it into the future with estimated growth rates for different economic activities. The model may also be more complicated and take into account interactions between different (economic) sectors and activities. For this type of forecast one often uses macroeconomic models, and preferably so-called 'computable general equilibrium' (CGE) models. This chapter will consider some examples of forecasting and policy analysis based on the second alternative, in the form of economic models, to see how they could be used together with information about material flows in order to forecast these flows.

When using models to do a forecast it is important to keep in mind that it will only give a picture of a possible development; it can never be looked upon as a definite answer. A forecast is most useful when comparing different possible developments, and especially in seeing how different policy measures might affect the development. One often starts with a business-as-usual path; that is, what is thought to be the most likely development given actual trends and today's policy. Then, by using the model, one constructs one or more new paths where the measure to be analyzed is implemented. Comparing these various paths gives a picture of how effective the measure might be.

It is also important to keep in mind that a model is necessarily a description of a limited aspect of a society. A model builder must always make a choice between simplicity and realism; the simpler (and maybe more user-friendly) the model, the less realistic. A model that is very detailed and hence more realistic runs the risk of being difficult to manage (although modern computer science has largely eliminated computational problems) and, worse, opaque. The more complicated the model is the more difficult it will be to interpret the results, and to determine how different effects interrelate. This is a crucial point, because it explains why most economic models up to now have neglected material/energy flows.

ECONOMIC MODELING

Economic models can be viewed as paper laboratories economists can use to conduct *gedanken* experiments, since it is impossible to perform these experiments in real life. There exist several types of models that can be used for different types of analyses of economic character, including macroeconomic, input–output and general equilibrium models. The models all have different virtues and drawbacks that cannot be further examined here. For long-term forecasts of the allocation of resources, that is labor, and all kind of physical materials, capital and consumption goods, one often uses general equilibrium models. Therefore we will focus mainly on this type of model.

The idea of general equilibrium is a fundamental pillar in economic theory, and basically it assumes that all the markets that make up an economy either are in or tends towards a state of equilibrium. This means that for each market the supply of each good or service will equal the demand for that good or service. Adam Smith's notion about the invisible hand coordinating market clearance can be viewed as the starting point of the theory of general equilibrium (Smith 1993 [1776]), but the first to formally describe general equilibrium was Walras (1995 [1874]). The idea was developed over the years and can be said to have reached maturity in the work by Arrow and Debreu (1954). It is therefore often referred to as the Arrow–Debreu economy. Introduction to the formal theory of general equilibrium can be found in numerous economic textbooks; see, for example, Hildenbrand and Kirman (1988), Ellickson (1993) and Myles (1995).

By the development of computable (that is, numerical multisectoral) general equilibrium models (CGE models), the theory of general equilibrium became an operational tool in empirically oriented economic analysis. CGE models with realistic empirical representation of one or more countries are often called applied general equilibrium (AGE) models. The models consist of a set of aggregated economic agents, who demand or supply aggregated goods (consumption goods, services and production factors). The agents are supposed to be rational in the economic sense, meaning that consumers maximize their utility and producers maximize profit. The model endogenously determines quantities and relative prices, at a point in time, and thereby the resource allocation in the economy, such that all markets clear.

It is commonly agreed that CGE modeling began with the work of Johansen (1960), which was the starting point for the MSG (multisectoral growth) model, a model still being developed and refined at Statistics Norway and used by the Norwegian authorities. The two latest versions, MSG-5 and MSG-6, are documented in Holmøy (1992) and Holmøy and Hœgeland (1997). The CGE model ORANI, developed for Australia, can be looked upon as an elaboration of the MSG model, see Dixon *et al.* (1982). Introduction to and surveys of the methods and theories developed in AGE and CGE modeling can be found in Fullerton *et al.* (1984), Shoven and Whalley (1984, 1992), Bovenberg (1985), Bergman (1990) and Dixon *et al.* (1992).

Of course there exists a lot of criticism of general equilibrium models, basically referring to the (lack of) realism in describing the functions of a society. The criticism ranges from the need to extend and further develop GE models, as in Walker (1997), to a more fundamental critique of the underlying theory, especially the lack of an endogenous theory of technological progress and the implausibility of growth in a state of static equilibrium. See, for example, Black (1995) and Ayres (2000).

INTEGRATED ECONOMY–ENVIRONMENT MODELS

Partly in the aftermath of the oil crisis in the mid-1970s, and especially as a response to the emerging environmental debate, including the risk of climatic changes, a great deal of the development of AGE/CGE models during the last decades has been aimed at energy and environmental issues. Sometimes these models are referred to as KLEM models; that is models where production is based on the production (or input) factors capital (*K*), labor (*L*), energy (*E*) and materials (*M*), the latter including both natural resources (raw materials) and intermediate goods. For a review of some of these models and a discussion of the integration of environmental concerns in economic models see Forssell (1998).

Inasmuch as the greenhouse gas issue and the risk of climatic changes is a global concern, some global models have been developed to study effects of different policies aimed at reducing man-made emissions of these gases. These models are also used to study different implementations of international agreements like the Kyoto protocol. One such model is GREEN (General Equilibrium Environment Model), an AGE model developed by the OECD Economics Department. A non-technical overview of this model is found in Burniaux *et al.* (1992). Another global general equilibrium model is the G-Cubed model; see McKibbin and Wilcoxen (1992) and McKibbin (1998).

Integrated environmental and economic AGE models have also been used to study so-called 'green' tax reforms. In these taxes are shifted from, for instance, labor to environmental problems like energy or emissions. The idea is that such a tax shift may (or may not) yield a double dividend; that is both reduced environmental pressure and increased welfare, for instance through reduced unemployment. Examples of such studies are Jorgenson and Wilcoxen (1994), Goulder (1995), Carraro (1996) and Bruvoll and Ibenholt (1998).

MODELS INTEGRATING MATERIAL FLOWS AND ECONOMIC CONCERNS

The main purpose of this chapter is to describe models that can be used to estimate possible trends in the development of total consumption of physical materials. So far there are not many documented studies in this field: the models used for economic and environmental forecasts most often have dealt with the costs of emissions to air, and are expressed in monetary instead of physical units. To forecast physical material flows, including emissions, one has to integrate an economic model that predicts future extraction, production and consumption with a model that estimates some physical measurements of materials and natural resources, preferably the weight in tons or the embodied exergy content of wastes (a measure of its potential to initiate a chemical reaction with the environment).

An early contribution by Ayres and Kneese (1969) points to the need to integrate a material balance perspective in economic modeling. This need is based on the fact that residuals (waste) are an inherent and normal part of production and consumption. Further, the quantities of these residuals increases with increases in population and/or level of output, and they cannot be properly dealt with by considering different environmental medias in isolation. Ayres and Kneese construct a formal theoretical extension of a general equilibrium model (the so-called 'Walras–Cassel' model), that includes the mass balance condition by introducing an (unpriced) environmental sector and using physical units for production and consumption. In order to become an analytical tool this model has to be fed with enormous amounts of data, and the computation, at least at that time, would have been extremely difficult. The theoretical model is still useful, however, as it shows that partial analysis of isolated environmental problems can lead to serious errors.

There were hardly any applications of the idea propagated by Ayres and Kneese (1969) until 1994, when a work was published describing a dynamic macroeconomic model with a material balance perspective (van den Bergh and Nijkamp 1994). The authors' aim was to construct a model suitable for studies of the long-term relationship between an economy and its natural environment. The model was designed to capture two main elements. The first was the two-way interaction between population growth, investments, technology and productivity, on one side, and declining environmental quality and resource extraction on the other. The second element was a more realistic representation of the interdependence between various environmental effects achieved by using the material balance perspective. The model integrates economic growth theory and material balance accounting by combining complex interactions between the economy and the environment. It is not analytically soluble, but is more suitable for simulation. It was calibrated to fulfill certain conditions in a base case scenario, in which logical, realistic or plausible values were chosen for different variables. Then 10 different scenarios were constructed, changing initial stocks of capital, natural resources, pollution and/or non-renewable resources, including or not including ethical concerns and feedback from environment to investment. Van den Bergh and Nijkamp concluded that cautious behavior regarding the environment in the long run does not necessarily lead to (strongly) declining economic performance.

Another effort to connect a material balance module to the MSG model is documented in Ibenholt (1998). The main purpose of that study was to analyze the generation of waste in production processes, on the basis of the physical law of conservation of mass. The difference between the physical input (raw materials and intermediate goods) and the produced physical output (intermediate or final goods) is the residual consisting of emissions to air, land and water. The MSG-EE model, an energy and environmental version of MSG (Alfsen *et al.* 1996), was used to predict the economic variables needed for the analysis, namely production and use of different physical inputs, all measured in monetary units. The factors converting monetary to physical units were assumed to be constant during the forecasting period (1993–2010), meaning that each monetary unit of a physical input or product in each production sector has a constant weight. This is of course a simplification, but it may be fairly realistic, considering the aggregation level (between 30 and 40 physical input and output goods). The method does not consider changes in the material intensity of each physical input or output, but it does incorporate changes in the amount of total material input per produced unit.

The study predicts a growth in the residuals from manufacturing industries of 74 per cent from 1993 to 2010. The growth is partly explained by an anticipated growth in material intensity along the economic development path. Increasing material intensity is partly caused by the strong substitution possibilities between labor and material input in the MSG model, and it might very well be overestimated. The study did not include any alternative scenarios, such as different policies towards the material consumption, since the main purpose was to compare the mass balance perspective on waste generation with the method used in Bruvoll and Ibenholt (1997), where the generation of waste was explained by the development either in physical input or in production.

Another approach is described in Dellink and Kandelaars (2000). They combined the Dutch AGE model Taxinc with the material flow model Flux, which is an input–output type of database that describes the physical flows of materials in the Netherlands in 1990. The integration is incomplete since there is no endogenous feedback between the two models. The purpose of the study is to analyze material policies with the aim of reducing the use of specific materials (zinc and lead). The following policies were simulated: a regulatory levy on the primary use of zinc, on the throughput of zinc, on products that contain zinc, on the primary use of lead, and on the primary use of both zinc and lead. The tax revenue from the material levy was redistributed by reducing the employer's contribution to social security. First the material flow model is used to determine the use of zinc and lead in different production sectors, which determines the magnitude of the tax for each sector. The levy is then imposed in the Taxinc model and a new equilibrium is calculated. The result from the Taxinc model is imported to the Flux model to calculate the effect in physical units. The conclusion that can be drawn from the study is that the macroeconomic impact of the tested tax policies achieved reductions in material use of 5 to 10 per cent while total production decreased by less than 0.2 per cent. The material-intensive production sectors would, however, suffer rather severe effects. Since the model does not allow for substitution between different materials, the results should be interpreted with great care.

Two related studies are Bruvoll (1998) and Bruvoll and Ibenholt (1998). Bruvoll (1998) uses the MSG model to simulate a green tax reform where a tax is levied on plastic, wood pulp, cardboard and virgin paper materials, while the payroll tax is decreased (employers' contribution to social security). The tax rate in different production sectors is calculated on the basis of data from the national accounting system. The effects from this tax reform are similar to the ones in Dellink and Kandelaars (2000), namely that a rather substantial reduction in material use is possible at a rather low macroeconomic cost. Bruvoll and Ibenholt (1998) levy a general tax on all materials used in production, and show a clear, positive environmental effect in the form of reduced emissions to air and waste quantities. However, the welfare effect is uncertain owing to reductions in production and material consumption.

Other studies that forecast waste generated are Nagelhout *et al.* (1990), Bruvoll and Ibenholt (1997, 1999) and Andersen *et al.* (1999). All these studies use fixed coefficients to explain the generation of waste, but they differ in the choice of explanatory variables. Nagelhout *et al.* (1990) and Andersen *et al.* (1999) use production and consumption forecasts by an economic model as explanatory variables, whereas Bruvoll and Ibenholt (1997, 1999) link waste generated in production sectors to the use of intermediates. A weakness of the method of fixed coefficients is the inability to capture changes in the material intensity that ought to lead to changing waste amounts.

In summary, there are few economic models integrating a material perspective and none of them can be regarded as anything more than a step towards a comprehensive and analytical model. Nevertheless, these models can yield valuable insights.

CONVERTING BETWEEN ECONOMICAL AND PHYSICAL DATA

Since an economic model uses monetary units, whereas a material flow analysis uses physical units, some link between these different units is needed. The increasing effort to construct physical materials accounts and to incorporate these in the monetary national accounts (see Chapters 8 and 10) will most certainly be of valuable help in this conversion. The model used in Dellink and Kandelaars (2000) uses data from a physical material account, whereas the model in Ibenholt (1998) uses a quite different approach and constructs conversion factors based on detailed manufacturing statistics. The choice of method was mainly due to the fact that it was only in these statistics that physical data were available for Norway for the base year (1993).

A common simplification in most models mentioned above is the assumption of constant conversion factors between physical and monetary units. The models disregard the fact that product development and/or changes in the composition of aggregated commodities can cause the mass of these commodities to change. Constructing exogenously variable conversion factors would probably not be technically difficult, but the problem lies in determining how these factors should develop over time. Endogenously variable conversion factors would be a far more difficult task, and certainly beyond current capabilities. If material accounts, or other forms of physical statistics, become more common and are constructed on a regular basis it would be possible, at least in theory, to construct time series of conversion factors. This could ultimately prove useful when determining how conversion factors develop over time, and what might affect them.

TECHNOLOGICAL DEVELOPMENT

A major weakness with general equilibrium models is the way they handle technological development. The most common way to specify technological progress in these models is to use so-called ‘Hicks-neutral’ progress within each sector; that is, annual efficiency gain is assumed equal for all factor inputs in the same sector. Thus it does not directly influence the relationship between the various factor inputs within a sector. However, it affects relative factor prices and thereby, indirectly, changes the composition of the factor inputs in a sector. This approach assumes that technological progress is exogenous. In addition, it is likely that most CGE models can only anticipate marginal changes in technological progress, since substantial changes might make the model (which assumes growth in perpetual equilibrium) collapse.

If one uses an input–output model for forecasts, instead of an AGE model, it is easier to apply larger shifts in the technological progress rate. A global input–output model for forecasts of global emissions of greenhouse gases for the years 2010 and 2020 is the World Model (Duchin and Lange 1996). Also several efforts have been made to endogenize technological development in economic models. This branch of economic thought is called the 'new' theory of endogenous growth; see, for instance, Romer (1986, 1987, 1990), Lucas (1988) and Grossman and Helpman (1994). For discussion of methods of endogenous technological progress and environmental issues, see Victor *et al.* (1994), Goulder and Schneider (1999) and Parry *et al.* (2000).

REBOUND

Despite the imperfect handling of technological development in AGE models, they offer valuable insight into one effect such development can have on the use of material. In the literature on energy use there has long been a discussion about the so-called 'rebound effect', meaning that more efficient energy equipment could increase total energy use. See *Energy Policy* (2000) for an overview and summary of this discussion.

In some cases, at least, increased efficiency in the utilization of resources in production may result in a fall in real prices of the commodities/resources that experience the strongest efficiency increases. This will make us richer (income effect) and at the same time physical resources and products become cheaper (price effect). Being richer we can consume more. Since physical goods become relatively cheaper we can increase the demand for them. Through these mechanisms increased resource efficiency might actually increase the total use of the resource. This is the case in the study by Ibenholt (1998), where rebound is one of the main causes of the strong growth in residuals. See also Chapter 18 for a discussion of rematerialization that might be due to this rebound effect.

PRICES

Commodities are essentially (transformed) natural resources with a rent attached. From a welfare perspective it is optimal to tax this rent, even though the tax base could be rather difficult to define in practice. Another argument for taxation of material consumption is the environmental problems this consumption causes. According to economic theory a cost-effective way to deal with such problems is through different forms of taxation. See, for instance, Pearce and Turner (1990), Repetto *et al.* (1992) and Lesser *et al.* (1997). The studies by Bruvoll (1998), Bruvoll and Ibenholt (1998) and Dellink and Kandelaars (2000) are all based on the idea of pricing materials in accordance with the environmental problems they incur, and they all show that this might be a rather cost-effective way to reduce environmental pressure.

The price mechanism is an important tool for steering technological development.

Increased prices on natural resources and materials will most certainly spur technological development towards less material-intensive products and production processes, while at the same time dampening the rebound effects of this development. Keeping real prices of the physical resources constant, or even letting them rise, would modify the demand-increasing (rebound) effect of technological progress. As mentioned above, ordinary AGE models do not fully capture the effect the price mechanism would have on technological development. This, in context, must be considered a severe weakness.

CONCLUSION

Even if there does not exist – and maybe never will exist – any fully integrated model, striving towards such a model gives us valuable insights about the economic forces steering material use and how this might affect the environment. Weaknesses yet to be fully addressed include the impact of possible future scarcity, as reflected in non-declining resource prices, and the negative impact of pollution on unpriced but essential environmental assets. In short, more needs to be done in terms of accounting for the impact of resource use on the economic system.

Up to now, forecasts of the consumption of different types of materials have been essential since many of the environmental and resource problems are rooted in this consumption. Total material use can also serve as an indicator of sustainable development. For example, Hinterberger *et al.* (1997) propose total material requirement (TMR) as a better indicator than ‘constant natural capital’. However, the risk in focusing on TMR is that it is easy to overlook small, but very damaging, material streams. For this purpose studies like Dellink and Kandelaars (2000) might serve better.

Despite their many weaknesses, general equilibrium models must still be regarded as useful tools for studying interactions between different sectors of an economy and the environment. There is, however, a need for dynamic multisectoral models that capture both the material balance perspective and endogenous technological progress.

16. Exergy flows in the economy: efficiency and dematerialization

Robert U. Ayres*

BACKGROUND

The possible contribution of natural resource inputs to growth (or to technical progress) was not considered seriously by mainstream economists until the 1970s (mainly in response to the Club of Rome and 'Limits to Growth'), and then only as a constraint (Dasgupta and Heal 1974, 1979; Solow 1974a; Stiglitz 1974, 1979). It follows that, in more recent applications of the standard theory (as articulated primarily by Solow), resource consumption has been treated as a consequence of growth and not as a factor of production. This simplistic assumption is built into virtually all textbooks and most of the large-scale models used for policy guidance by governments.

The reality is considerably more complex. Looking at the growth process itself, it is easy to see that there are several identifiable 'growth engines' that have contributed to economic growth in the past, and still do, albeit in variable combinations. A 'growth engine' is a positive feedback loop or cycle. In Marshallian neoclassical economic theory, increased demand generates increased supply through savings by capitalists and investment in new capacity. More consumers and more workers led to greater aggregate income, larger savings pools and more investment. The reverse part of the feedback cycle was based on 'Say's law', namely the proposition that 'supply creates its own demand' through declining prices (or increased quality) of products and consequent increasing demand for products (now expressed as price elasticity of demand). However, the savings and investment part of the feedback loop, in particular, is inadequate to explain what has happened since the beginning of the industrial revolution.

The real 'growth engine' of the first industrial revolution was the substitution of coal for charcoal from wood and the development of steam power. The positive feedback cycle operated through rapidly declining fossil fuel and mechanical power costs, and their relationship with scale of production, on the one hand, and demand for end-use products, on the other. The growth impetus due to fossil fuel discoveries and applications continued through the 19th century and into the 20th with petroleum, internal combustion engines, and – most potent of all – electrification. The advent of cheap electricity in unlimited quantities has triggered the development of a whole range of new products and industries, including electric light, radio and television, moving pictures, and new materials, such as aluminum and superalloys, without which the aircraft and aerospace sectors could not exist.

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In effect, energy consumption within the economy is as much a driver of growth as a consequence of growth. It is also a very plausible surrogate for technological change, in Solow's sense. The point is that, to a naive observer, energy and material resources are as much a factor of production as labor or capital. Moreover, it is entirely plausible that resource consumption is a reasonable proxy for technical change, or 'technological progress' in Solow's theory. If so, it follows that one can construct a theory of growth that is endogenous, that is, in which there is no need for an exogenous driving force. This 'new' growth theory would, incidentally, constitute a strong link between industrial metabolism or industrial ecology and conventional economic ideas.

EXERGY – A USEFUL CONCEPT

As mentioned above, there exist several identifiable 'engines of growth' (positive feedback cycles) of which the first, historically, and still one of the most powerful, has been the continuously declining real price of physical resources, especially energy (and power) delivered at a point of use. The tendency of virtually all raw material and fuel costs to decline over time (lumber was the main exception) has been thoroughly documented, especially by economists at Resources For the Future (RFF). The landmark publication in this field was the book *Scarcity and Growth* (Barnett and Morse 1963), updated by Barnett (1979). The details of historical price series, up to the mid-1960s, can be found in Potter and Christy (1968). The immediate conclusion from those empirical results was that scarcity was not in prospect and was unlikely to inhibit economic growth in the (then) foreseeable future. It is also very likely, however, that increasing availability and declining costs of energy (and other raw materials) has been a significant driver of past economic growth. The increasing availability of energy from fossil fuels has clearly played a fundamental role in growth since the first industrial revolution. Machines powered by fossil energy have gradually displaced animals, wind power, water power and human muscles and thus made human workers vastly more productive than they would otherwise have been.

The word 'energy' in the previous paragraph is commonly understood to mean 'available energy' or 'energy that can be used to do work' in the technical sense. However, the first law of thermodynamics in physics is that energy is conserved. The total energy in a system is the same before and after any process. It is not energy, per se, but 'available energy' that can 'do work' or drive a process of transformation. The accepted thermodynamic term for this quantity is *exergy*. Exergy is not conserved. On the contrary, it is 'used up' (and converted, so to speak, into entropy).

The technical definition of exergy is the maximum amount of work that can be done by a system (or subsystem) approaching thermodynamic equilibrium with its surroundings by reversible processes. The equilibrium state is one in which there are no gradients: energy, pressure, density and chemical composition are uniform everywhere. The term 'work' here is a generalization of the usual meaning. For example, a gas consisting of one sort of molecules diffusing into a gas consisting of another sort of molecules (for instance, carbon dioxide diffusing into the air) 'does work', even though that work cannot be utilized for human purposes. Nevertheless, exergy is a measure of distance from equilibrium, and the important point is that all materials – whether they are combustible or not – contain some exergy, insofar as they have a composition different from the composition of the surrounding reference system. (For more detail see any suitable thermodynamics text, such as Szargut *et al.* (1988). Iron ore contains exergy, for instance, because it contains a higher proportion of iron and a lower proportion of silica and alumina and other things than the earth's crust. Carbon dioxide contains some exergy precisely because it differs chemically from the average composition of the earth's atmosphere. The exergy content of a non-combustible substance can be interpreted (roughly) as the amount of fuel exergy that would have been required to achieve that degree of differentiation from the reference state. On the other hand, all combustible substances, especially fossil fuels, have exergy contents only slightly different from their heat values (known as enthalpy).

In short, virtually all physical substances – combustible or not – contain exergy. Moreover, the exergy of any material can be calculated by means of precise rules, as soon as the surroundings (that is the reference state) are specified. From a biological–ecological perspective, solar exergy is the ultimate source of all life on earth, and therefore the source of economic value. This idea was first proposed by the Nobel laureate chemist Frederick Soddy (1922, 1933) and revived by the ecologist Howard Odum (1971, 1973, 1977), and economist Nicholas Georgescu-Roegen (1971, 1976b). A number of attempts to justify this bioeconomic or biophysical view of the economy by econometric methods using empirical data followed (Costanza 1980, 1982; Hannon and Joyce 1981; Cleveland *et al.* 1984).

However, despite the impressively close correlations between gross exergy consumption and macroeconomic activity as revealed by the work of the biophysical group cited above, the underlying energy (exergy) theory of value is impossible to justify at the microeconomic level and it is quite at odds with the paradigm of mainstream economics which is built on a theory of human preferences (for example, Debreu 1959). There will be comment further on this point later.

Nevertheless, exergy analysis has its uses. Exergy is a general measure applicable to all material resources at any stage of processing, including minerals and pollutants. It can be applied to the evaluation and comparison of resource availability (for example Wall 1977). From a theoretical perspective, the economic system can be viewed as a system of exergy flows, subject to constraints (including the laws of thermodynamics, but also others) and the objective of economic activity can be interpreted as a constrained value maximization problem (or its dual, an exergy minimization problem) with value otherwise defined (Eriksson 1984). Exergy analysis can also be used empirically as a measure of sustainability, to evaluate and compare wastes and emissions from period to period or country to country (Ayres *et al.* 1998). Reference is made to exergy, hereafter, even where the word 'energy' is used in its familiar sense.

THE ROLE OF EXERGY IN GROWTH

The generic exergy-driven positive feedback growth cycle works as follows: cheaper exergy and power (due to discoveries, economies of scale and technical progress in energy conversion) enable goods and services to be produced and delivered at lower cost. This is another way of saying that exergy flows are 'productive'. Lower cost, in competitive markets, translates into lower prices which – thanks to price elasticity – encourage higher demand. Since demand for final goods and services necessarily corresponds to the sum of factor payments, most of which flow back to labor as wages and salaries, it follows that wages of labor and returns to capital tend to increase as output rises. This, in turn, stimulates the further substitution of fossil energy and mechanical power for human (and animal) labor, resulting in further increases in scale and still lower costs. The general version of this feedback cycle is shown schematically in Figure 16.1.

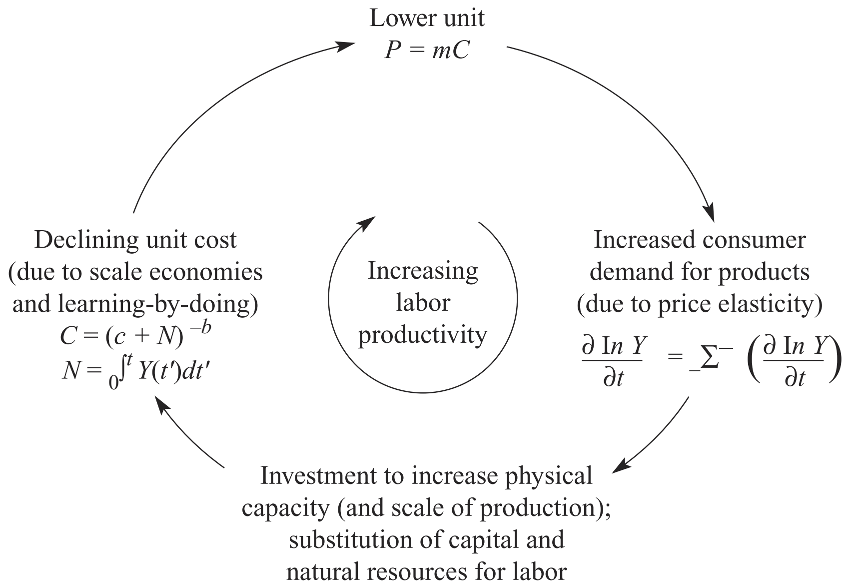


Figure 16.1 The Salter cycle growth engine

Marx believed (with some justification at the time he wrote) that the gains would flow mainly to owners of capital rather than to workers. Political developments have changed the balance of power since Marx's time. The division between labor share and capital share has been remarkably constant over many decades, although the capital share has been increasing in recent years. However, whether the gains are captured by labor or capital does not matter: in either case, returns to energy (or natural resources) decline as output grows. This can be interpreted as a declining real price.

Based on both qualitative and quantitative evidence, the positive feedback relationships sketched above imply that physical resource flows have been, and still remain, a major factor of production. It is not surprising, therefore, that including a resource flow proxy in the neoclassical production function, without any exogenous time-dependent term, seems to account for economic growth quite accurately for significant time periods, as noted above.

Among many neoclassical economists, strong doubts remain. It appears that there are two reasons. The first and more important is theoretical: national accounts are set up to reflect payments to labor (wages, salaries) and capital owners (rents, royalties, interest, dividends). In fact, GDP is the sum of all such payments to individuals. If labor and capital are the only two factors, neoclassical economic theory asserts that the productivity of a factor of production must be proportional to the share of that factor in the national income. This proposition gives the national accounts a fundamental role in production theory, which is intuitively attractive.

As it happens, labor gets the lion's share of payments in the national accounts, around 70 per cent, and capital (that is interest, dividends, rents and royalties) gets all of the rest. The figures vary slightly from year to year, but they have been relatively stable (in the USA) for most of the past century. Land rents are negligible. Payments for fossil fuels (even in 'finished' form, including electric power) altogether amount to only a few per cent of the total GDP. It seems to follow, according to the received economic theory of income allocation, that exergy and natural resources are not a significant factor of production and can be safely ignored.

Of course, there is an immediate objection to this line of reasoning. Suppose there exists an unpaid factor, such as environmental services? Since there are no economic agents (that is persons or firms) who receive money income in exchange for environmental services, there are no payments for such services in the national accounts. Absent such payments, it would seem to follow from the above logic that environmental services are not economically productive. This implication is obviously unreasonable. In fact, it is absurd.

The importance of environmental services to the production of economic goods and services is difficult to quantify in monetary terms, but conceptually that is a separate issue. Even if such services could be valued very accurately, they still do not appear directly in the national accounts and the hypothetical producers of economic goods would not have to pay for them, as such. There are some payments in the form of government expenditures for environmental protection, and private contributions to environmental organizations, but these payments are counted as returns to labor. Moreover, given the deteriorating state of the environment, it seems clear that the existing level of such payments is considerably too low. By the same token, the destruction of unreplaced environmental capital should be reflected as a deduction from total capital stock for much the same reasons as investments in reproducible capital are regarded as additions to capital stock.

Quite apart from the question of under-pricing, the apparent inconsistency between very small factor payments directly attributable to physical resources – especially energy – and very high correlation between energy inputs and aggregate economic outputs can be traced to an often forgotten simplification in the traditional theory of income allocation. In reality, the economy produces final products from a chain of intermediates, not directly from raw materials or, still less, from abstract labor and abstract capital.

Correcting for the omission of intermediates by introducing even a two-sector or three-sector production process changes the picture completely. In effect, downstream value-added stages act as productivity multipliers. Or, to put it another way, the primary sector can be considered as an independent economy, producing value from inputs of physical resources and small inputs of labor and capital. The secondary sector (or economy) imports *processed* materials from the first sector and uses more labor and capital (and processed materials) to produce still higher value products, and so forth. Value is added to materials step by step to the end of the chain. This enables a factor receiving a very small share of the national income, or even none at all, to contribute a much larger effective share of the value of aggregate final production. By the same token, this factor can be much more productive than its share of overall labor and capital would seem to imply.

The second source of doubt about the importance of resource consumption as a growth driver arises from the fact that even a high degree of correlation does not necessarily imply causation. In other words, the fact that economic growth tends to be very closely correlated with energy (exergy) consumption – a fact that is easily demonstrated – does not a priori mean that energy consumption is the cause of the growth. Indeed, most economic models assume the opposite: that economic growth is responsible for increasing energy consumption. This automatically guarantees correlation. It is also conceivable that both consumption and growth are simultaneously caused by some third factor. The direction of causality must evidently be determined empirically by other means.

There are statistical approaches to addressing the causality issue. For instance, Granger and others have developed statistical tests that can provide some clues as to which is cause and which is effect (Granger 1969; Sims 1972). These tests have been applied to the present question (that is, whether energy consumption is a cause or an effect of economic growth) by Stern (Stern 1993; see also Kaufmann 1995). In brief, the conclusions depend upon whether energy is measured in terms of heat value of all fuels (in which case the direction of causation is ambiguous) or whether the energy aggregate is adjusted to reflect the quality (or, more accurately, the price or productivity) of each fuel in the mix. In the latter case the econometric evidence seems to confirm the qualitative conclusion that energy (exergy) consumption is a cause of growth. Both results are consistent with the notion of mutual causation.

EXERGY AS A FACTOR OF PRODUCTION?

It is interesting to consider a relationship of the form:

$$Y = fgE \tag{16.1}$$

where Y is GNP, measured in dollars, E is a measure of 'raw' physical resource inputs (in exergy terms), f is the ratio of 'finished' exergy output to 'raw' exergy input, and g is the ratio of final output in money terms to finished exergy. There is no approximation involved in this formulation. It is an identity.

The virtue of this identity is that the terms can be given physical interpretations quite easily. Under certain conditions, discussed below, it will be seen that (16.1) can also be interpreted as a production function. A production function is a construct which attempts to explain economic activity (production) in terms of so-called 'factors of production'. These factors are supposed to be independent inputs that can be supplied in arbitrary proportions, such that a given output can be generated by a wide range of combinations of the inputs. In effect, the inputs are supposed to be substitutes for one another. In reality, none of the factors is independent of the others. Capital is unproductive without labor and exergy inputs. Similarly, labor produces nothing without capital (originally land and natural capital) and exergy inputs. Finally, exergy produces nothing without capital and labor.

An aggregate production function should also have the following features: (a) it should be consistent with the multi-sectoral (chain) model; (b) it should satisfy the usual condition of constant returns-to-scale; (c) all factor productivities should be positive and (d) it should replicate long-term economic growth of GDP reasonably well without introducing an exogenous 'technological progress' multiplier, using only capital, labor and 'energy' (actually exergy) as factors, and with the fewest possible independent parameters.

EXERGY EFFICIENCY AND WASTE

The definition of exergy efficiency f is somewhat arbitrary. We could draw the line (between f and g) in several ways. However, the most convenient division, partly for reasons of data availability, is the following. Finished exergy, the numerator of f consists of three components, namely physiological work by humans and farm animals, mechanical work by prime movers (internal combustion engines of all kinds and electric power produced by any means) and chemical exergy (heat) produced for any purpose other than driving an engine, including driving chemical reactions. The chemical exergy embodied in finished materials (contained in structures and durable goods) is almost entirely derived from fuels or electric power. The exergy contribution from metal ores is small and mostly attributable to sulfur in sulfide ores, which can be lumped with fuels.

The above definition omits end-use efficiency, the efficiency with which heat or electric power delivered to a user is converted, within the service sector or within the household, into the ultimate service (climate control, cooking, washing, information processing or communication). This omission is unfortunate, since most of the technological progress in recent years, and most of the efficiency gains, have been in this area. Nevertheless, it is conceptually useful to distinguish the efficiency with which 'raw' exergy is converted into 'finished' exergy, consisting of work or heat delivered to a user and chemical exergy embodied in finished materials.

On the input side, exergy consists of the following: products of photosynthesis (phytomass), fossil fuels, nuclear heat, hydroelectric power, and metal ores and other minerals. Photosynthetic exergy utilization in the USA in 1998 consisted of primary agricultural phytomass generated for the food system (including grazing animals) – 24.5 exaJoules (EJ) – plus a small contribution by non-food crops (mainly cotton) plus wood. This analysis was made with the aid of an extremely comprehensive agricultural model (Wirsenius 2000) using FAO data for the years 1992–1994. The model works back from final food intake to primary production requirements, adjusting for trade. Food eaten in the USA itself amounted to just about 1 EJ, and exports increased this to 1.37 EJ (equivalent). The calculated efficiency of the US production system was 5.6 per cent, implying gross primary production of 24.5 EJ. Of this 15.6 EJ was actually utilized (harvested and processed or fed to animals), the remainder being unharvested, lost or used for other purposes such as seed, mulch or fuel. Roundwood harvested (for lumber and paper) accounted for an additional 4.85 EJ. Fuelwood is lumped with fossil fuels, which amounted to 80.9 EJ in 1998. Nuclear reactor heat added 7.3 EJ and hydroelectric power added 1.1 EJ. So-called 'energy' inputs altogether accounted for 89.3 EJ. Finally, the exergy value of metal ores added 0.41 EJ. The grand total of all inputs was slightly more than 114.8 EJ. This is the denominator of the efficiency ratio, f as of 1998.

The 'finished exergy' output components are mechanical work (done by humans, animals and machines), useful heat and materials. These can be evaluated numerically with modest effort and some reasonable assumptions. Animal and human work can be calculated from the caloric value of metabolizable food consumption, adjusted for metabolic efficiency and fractional working time. In the USA for the year 1998, farm animals did too little work to be counted. Humans in the USA consumed 1.0 EJ of food (caloric value) in 1998. Food consumption by 270 million people, at 3300 Cal/day, amounts to just over 1 EJ. However, men in the USA (and Europe) spend less than 20 per cent of lifetime 'disposable' hours doing work for pay (that is within the economic system). As regards women – because they live longer and spend somewhat less time doing paid work – the figure is probably around 15 per cent. Assuming very little exergy is needed during 'non-disposable' hours (sleeping, eating, personal hygiene and so on), the overall average fraction of exergy consumption devoted to economically productive work is still not above 15 per cent, at present. (It was probably twice that in 1900, however, when people worked many more hours and died younger.) The muscular efficiency of the human body is about 20 per cent. The product of the two efficiencies is less than 0.03; that is to say, human labor amounted to less than 0.03 EJ which is negligible. The details do not matter, since the absolute number is so small.

Prime movers (mostly car, truck, bus and aircraft engines) not used for electricity generation consumed 26.6 EJ, but the mechanical work done amounted to about 6.4 EJ (assuming 25 per cent net efficiency, after allowing for internal losses in vehicle drive trains and other parasitic loads). Net electric power output delivered was 13.3 EJ in 1998.

Finally, chemical exergy supplied for other purposes (mostly heat) consumed 33.6 EJ in fuel terms. The 'conversion' efficiency in this heterogeneous commercial–household–industry sector is difficult to estimate, since it includes space heat, domestic cooking and washing, and all kinds of chemical and metallurgical reduction and transformation processes driven by chemical (and, to some extent, electrical) exergy. The space heating, water heating and cooking component is especially difficult to evaluate, since end-use efficiencies in this area are extremely low in the 'second law' sense (that is, in comparison with the minimum amount of exergy required in principle by the most efficient possible way of delivering the same services. A 10 per cent figure for fuel is probably optimistic. The efficiency of most metallurgical and chemical processes (measured as exergy embodied in final products to exergy of fuel consumed) is somewhat higher, probably on average closer to 30 per cent. Combining the two, very roughly indeed, one might assume an average 15 per cent conversion efficiency. More precision is impossible without a detailed process-by-process analysis. On this basis, the 'output' in 1998 amounted to something like 5 EJ. It is important to remember that most of the net exergy 'content' of asphalt, plastics and metals (around 4 EJ) is mostly derived from fossil fuels (or electric power), so it is already included. Adding wood and paper products, the exergy efficiency of the US economy for 1998 was of the order of 27/115 (23 per cent) plus or minus 2 or so.

A similar calculation for 1900 can be carried out, albeit a little less accurately. In that year the primary agricultural biomass was probably about 20 per cent less than that for 1993. The argument is that the total amount of land devoted to agriculture in the USA has changed very little since 1900; land made more productive by irrigation (mainly in California and the southwest) is roughly balanced by land lost to agriculture in the southeast and northeast as a result of extensive erosion and urbanization. Elsewhere, as in the great plains, irrigation is mainly compensation for falling water tables. Similarly, the net impact of fertilizers is largely to replace nutrients lost to harvesting and topsoil erosion. Increases in net food production can be attributed mainly to reduced need for animal feed (for horses and mules), improved seeds (yielding more grain or other useful product per unit of phytomass), animal breeding (more milk or eggs per unit of feed) and reduced losses to insects, rodents and other pests.

Other inputs were from fossil fuels and fuelwood (8.92 EJ), and timber (2.04 EJ), for a total of about 31 EJ. The exergy outputs included mechanical work on farms done by horses and mules, which was about 0.22 EJ in 1920, and 20 per cent lower than that in 1900. In the 1920s, land needed to provide feed for horses and mules amounted to 28 per cent of total agricultural land in the USA (US Census 1975). Assuming this figure applied in 1918, the peak year (when the horse and mule population was 26 723 000), it would appear that gross primary production of the order of 7 EJ was required to feed horses and mules. It has been estimated that these animals required 33 units of food energy to produce one unit of work. On this basis, the net work output of farm animals would have been about 0.22 EJ, with an uncertainty of at least 20 per cent. For comparison, Hayami and Ruttan (1971) estimated that 6 EJ was used by the food system around 1920 for both animal feed and fuelwood. This would imply a somewhat lower share for animals. For comparison, by 1960, when tractors had essentially replaced farm animals, machines and chemicals consumed about 5 EJ of fuel (Steinhart and Steinhart 1974). Assuming 15 per cent net efficiency (high), this would have been equivalent to 0.75 EJ of net work. However, it is probable that farmers did more physical work in 1960, thanks to the availability of mechanization, than they would have done with animals in 1920, simply because machines are faster and require much less human labor.

There was a similar contribution by railroads and stationary engines in 1900, around 0.18 EJ (assuming 10 per cent thermal efficiency of the steam engines in use at the time). Other fuel use, consisting of domestic heat, process heat and exergy embodied in wood for construction and paper, might have been as large as 3 or 3.5 EJ. Adding them up, the overall exergy conversion efficiency in 1900 was probably less than 3.9/34.4 = 11.3 per cent, also plus or minus 2 per cent or around half of present levels. In short, f has been increasing, albeit at a modest rate, as indicated in Figure 16.2.

The exergy embodied in raw materials but not embodied in finished materials is, of course, lost as waste heat or waste materials (pollution), denoted W. All exergy converted to

$$f = \frac{E - W}{E} = 1 - \frac{W}{E} \tag{16.2}$$

heat or work is ultimately lost, of course. In fact, where both f and W can be regarded as functions of the three assumed factors of production, K, L, E. The exergy embodied in fuels and durable materials for the USA since 1900 is plotted in Figure 16.3.

It is noteworthy that, thanks to the increasing mechanization and electrification of the economy, the fraction of input exergy devoted to mechanical work has been increasing over time, whereas the fraction devoted to space heat and chemical work has been decreasing. The fraction devoted to powering 'prime movers' in the USA, from 1900 to 1998, is plotted in Figure 16.4.

On the other hand conversion losses associated with electric power generation and mechanical work are rather large, even after a century of improvement. The increasing efficiency of prime movers over the past century does not outweigh the increased demand for mechanical work (as electric power and transport). The case of electric power illustrates the point. The fuel required to generate a kilowatt-hour of electric

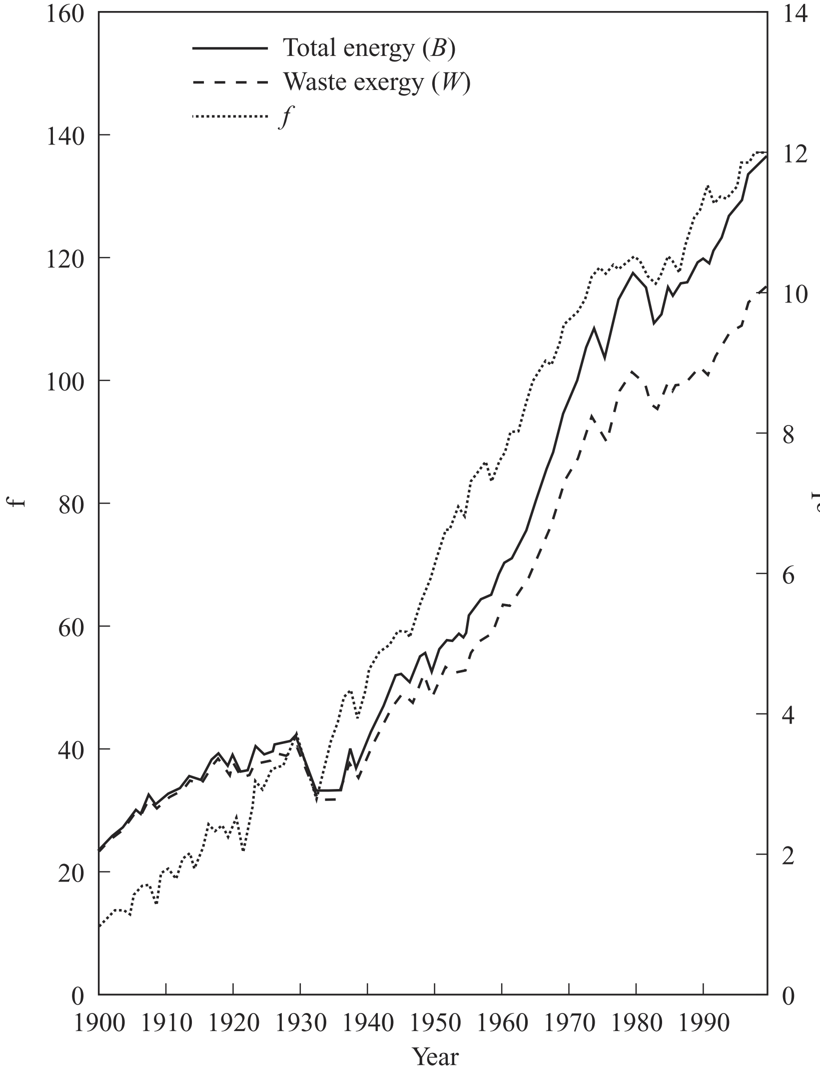


Figure 16.2 The ratio f *plotted together with* B*, total exergy and* W*, waste exergy – USA, 1900–1998*

power has decreased by at least a factor of six during the past century. On the other hand, the consumption of electricity in the USA has increased over the same period by more than a factor of 1000, as shown in Figure 16.5. For this reason, even though the conversion losses per unit of mechanical work have declined, this has mainly resulted in decreasing costs and increasing demand. However overall exergy conversion losses are increasing rapidly. This exemplifies the so-called 'rebound effect'. It is easy to see that the feedback growth mechanism illustrated in Figure 16.1 inherently depends upon this phenomenon. Without such a rebound effect it would be very difficult to sustain economic growth.

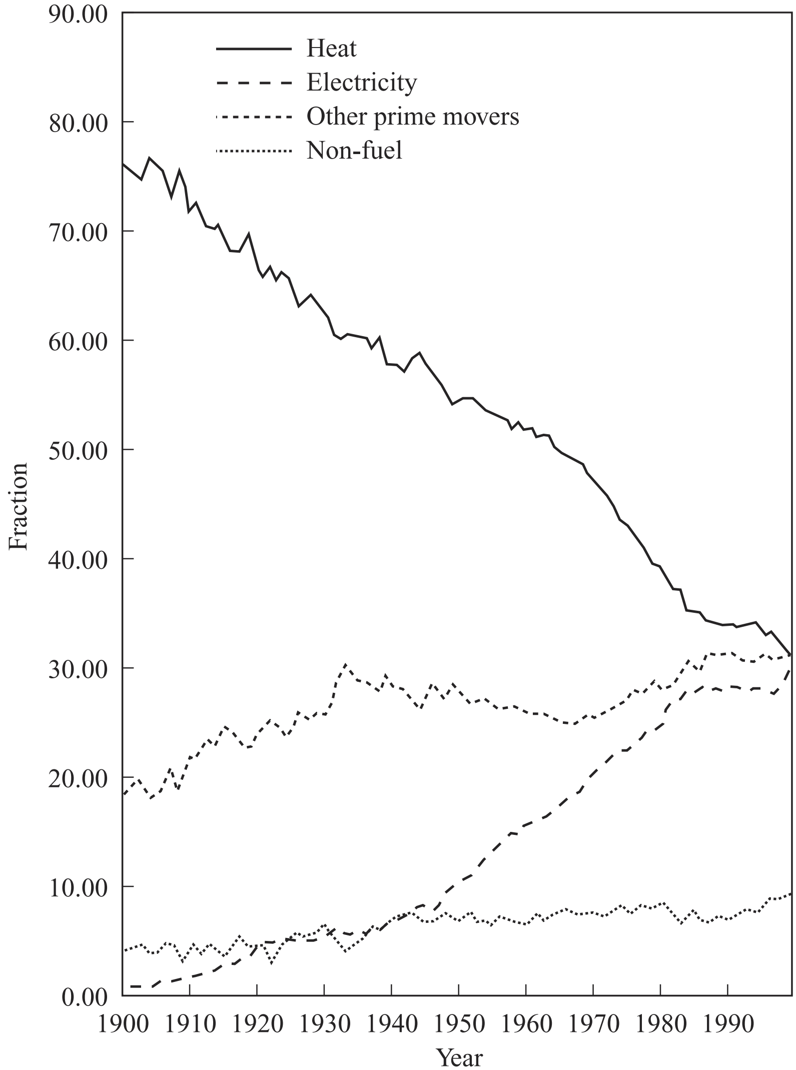


Figure 16.3 Fuel exergy used for different purposes – USA, 1900–1998

TECHNOLOGICAL CHANGE AND ENDOGENOUS ECONOMIC GROWTH

Returning to the issue of production functions, the 'chain' requirement (a) is not satisfied by a product of single sector production functions. This is because the constant returns requirement must apply to the whole chain. The output of the first (extractive) sector is only one of the inputs to the downstream sectors. Additional inputs of labor and capital are also needed to add value. The expression (16.1) above is consistent with the chain requirement (a) because fE can be interpreted as the physical output of the extraction and

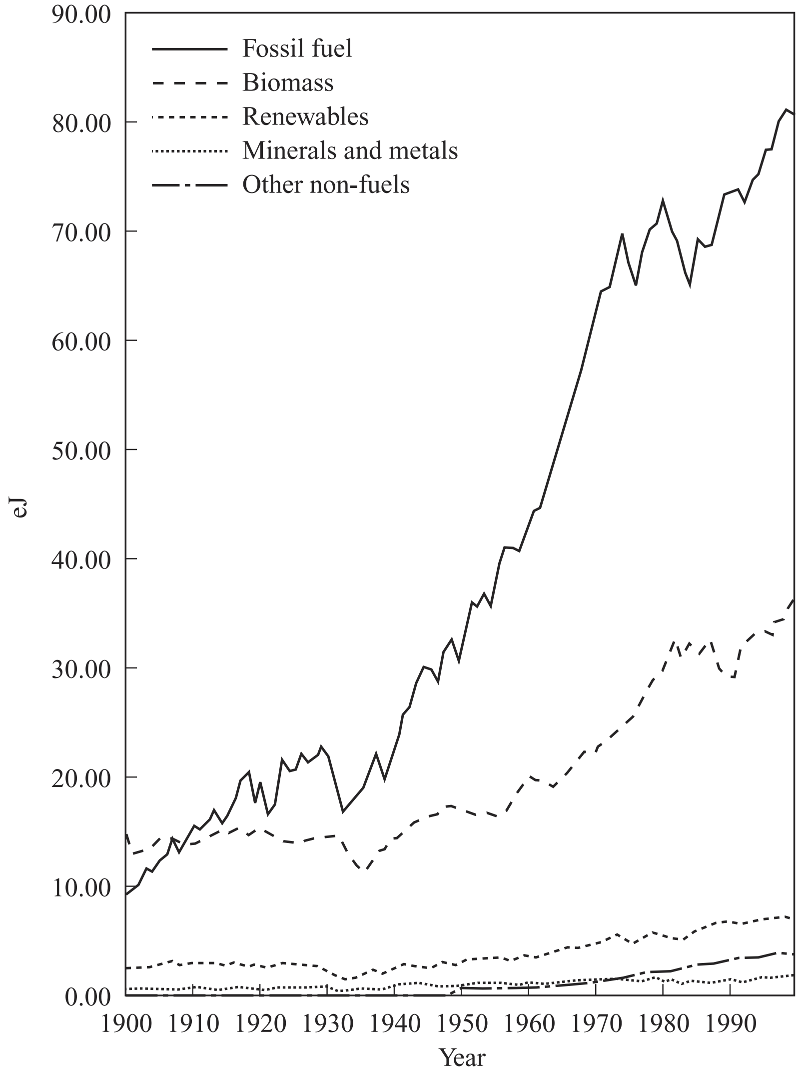


Figure 16.4 Breakdown of total exergy inputs – USA, 1900–1998

primary processing sector, while *g* expresses the combined value-added by any number of subsequent downstream sectors.

According to requirement (b) the product fgE must satisfy the Euler condition: it must be a homogeneous first order function of the variables, K, L and E. Since the term E is already first order, the Euler condition holds only if the product fg is homogeneous and of zeroth order in the same three production factors, K, L and E. The condition is satisfied by any function of simple ratios of the variables. (On the other hand we do not want fg to be an explicit function of time t. If t is not an independent variable the production function corresponds to an endogenous growth theory.)

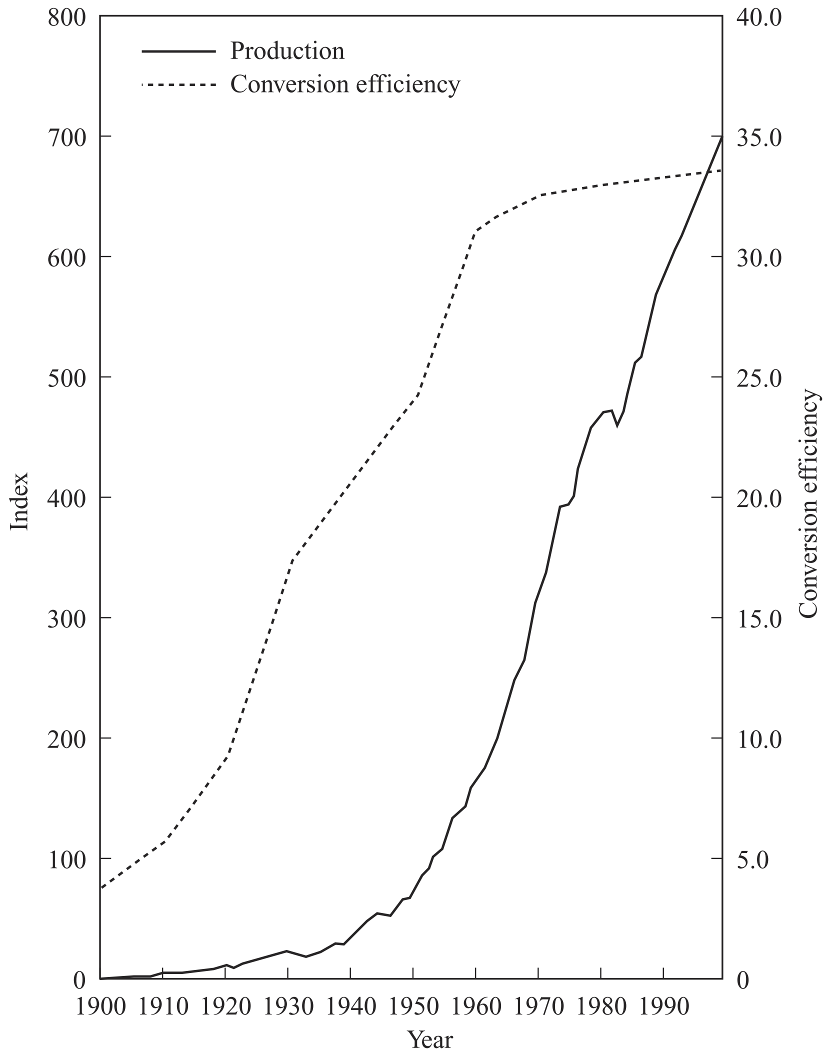


Figure 16.5 Index of total electrcity production by electric utilities (1900 = 1) and average energy conversion efficiency over time – USA, 1900–1998

Having calculated *f* we can now calculate *g*, from historical GDP data (in constant dollars). The result, GDP in 1992 dollars per unit of 'finished exergy' shows marked peaks during periods of upheaval such as wars and the Depression. The general trend declined during the first half of the century, but increased almost exponentially from 1960. The function *g* can be interpreted as a rough measure of the 'dematerialization' of the economy, in the very broad sense (counting fuels as materials). However, a more defensible measure of the material intensity of the economy is the ratio of exergy embodied in materials to the total exergy input to the economy. This ratio, although on a completely different scale, follows essentially the same pattern, although the impact of wartime

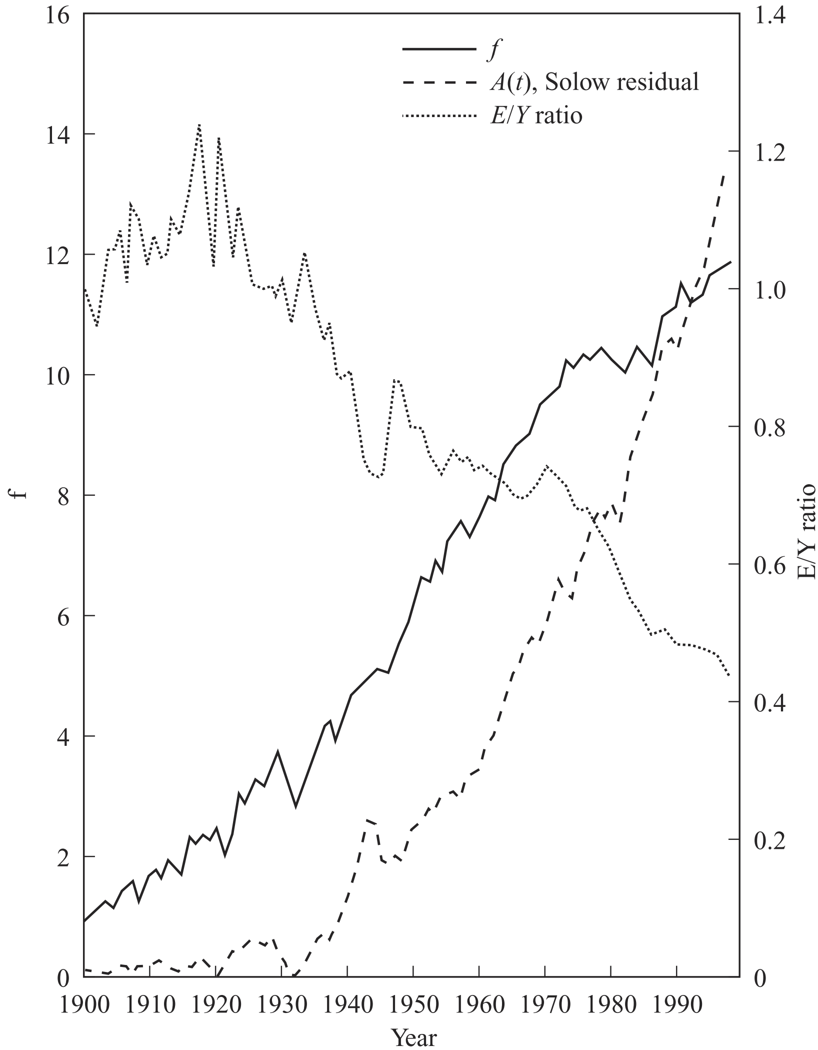


*Figure 16.6 Exergy intensity (*E/Y*) plotted against* f *and the Solow residual,* A(t) *– USA, 1900–1998*

stringencies is not apparent in this measure. Both ratios, on their own respective scales, are shown together in Figure 16.6.

The next challenge is to explain *f* and *g* in terms of the three independent production factors of production, *K*, *L*, *E* (excluding time), insofar as this is possible. The positive factor productivity requirement (c) and the 'good fit' requirement (d) must now be addressed together. It is not necessary that a simple functional form should satisfy the productivity conditions for all possible values of the variables. However, it is necessary that the conditions be satisfied for the range of values that have existed historically. Typically both K and L increase over time, but K normally increases faster (if there is growth) and E increases at an intermediate rate.

However, thanks to technical progress, GNP increases faster than any of the input factors, including E, although not as fast as mechanical work, especially electrical power output, has increased. Evidently both f and g and the product fg must also be increasing in the long run (though short-term fluctuations are not excluded).

It is more difficult than one might suppose to find functional forms that satisfy the combined requirements of constant returns, positive factor productivity, and endogeneity (that is without introducing a time-dependent multiplier). The last requirement alone rules out any Cobb–Douglas functional form $Y = AK^{a}L^{b}E^{1-a-b}$. This is because actual economic output Y has always grown faster than any of the individual input factors (K, L, E), and therefore faster than any product of powers of the inputs with exponents adding up to unity. (The 'best' fit is actually obtained by choosing $a = 1$ and $b = 0$, whence $Y = AK$, but even in this case A must be a function of time if constant returns to scale are required).

For purposes of illustration, Figure 16.7 shows the familiar Cobb–Douglas function with $A = \text{constant} = 1$ and exponents $a = 0.26$ and $b = 0.7$ (based on capital and labor shares of the national income). Obviously economic growth far outstrips the growth of the traditional factors K and L; the GDP 1900-based index is over 21, while K is under 11. For this case, the technology multiplier $A(t)$ can be fitted roughly for the entire period 1900–1995 by an exponential function of time (interpreted as a rate of technical progress) increasing at the average rate of 1.6 per cent per annum (Figure 16.8). This is similar to Solow's original result.

However, there are other functional forms combining the factors K, L, E that reduce the need for a time dependent multiplier, $A(t)$. As noted already, the form (6.1) can serve the purpose provided the argument(s) of f and g are increasing ratios of the factor inputs, such as K/L or E/L. It happens that a suitable functional form (the so-called LINEX function) has been suggested by Kümmel (Kümmel 1982a, 1982b; Kümmel *et al.* 1985):

$$Y = A\ E \exp\{aL/E - b(E + L)/K \tag{16.3}$$

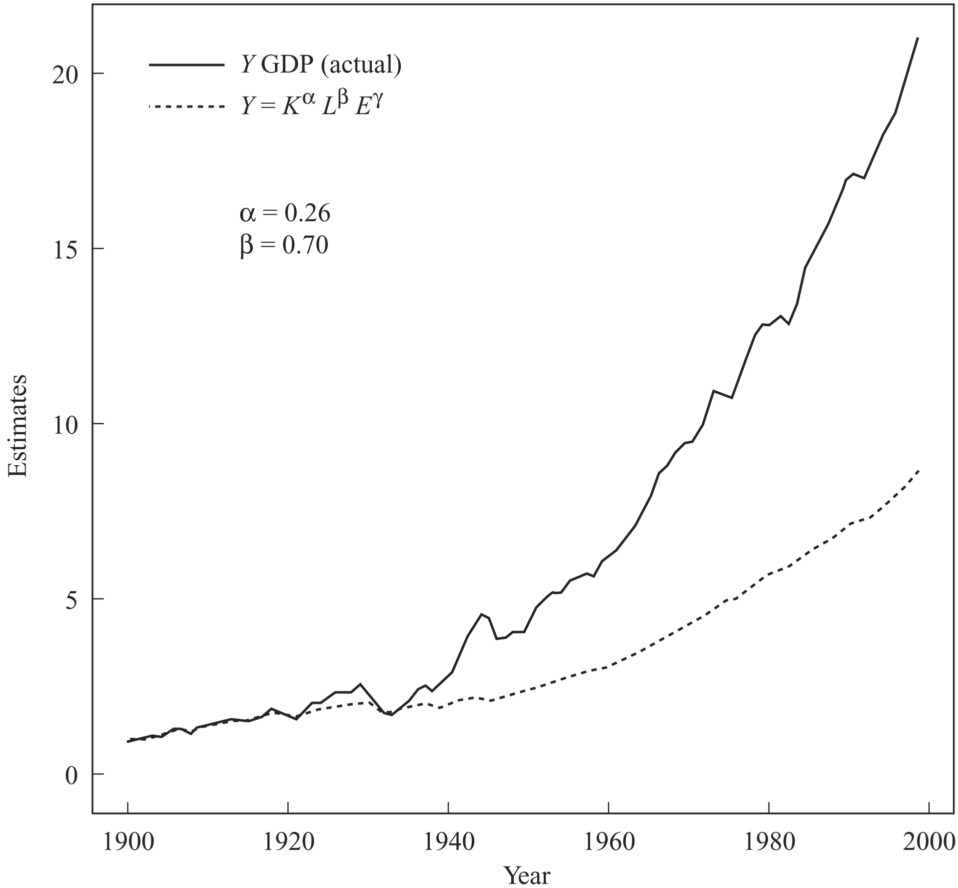
It can be verified without difficulty that this is a homogeneous function of the first order which satisfies the Euler condition for constant returns to scale. However, the requirement of positive factor productivity for all three factors is more difficult to satisfy with just two parameters. It can be shown that this requirement is equivalent to the following three inequalities:

$$b = 0 \tag{16.3a}$$

$$a > b\frac{E}{K} \tag{16.3b}$$

$$1 > a\frac{L}{E} + b\frac{E}{K} \tag{16.3c}$$

The first condition is trivial. The third can be rearranged. Introducing (16.3b) in (16.3c) one obtains:



Base year 1900 = 1992 $354 billion

Figure 16.7 Cobb–Douglas production function, USA, 1900–1998

$$1 > b\,\frac{E}{K}\left(1 + \frac{L}{E}\right) \tag{16.3d}$$

$$0 < b < \frac{K}{L + E} \tag{16.3e}$$

Obviously these conditions can be satisfied for all values of K, L, E within a reasonable range, but not for all possible ranges. The multiplier A, being time-independent, can be normalized to unity, with $Y(1995)$ set equal to 1.

A 'best fit' for a, b obviously imposes restrictions on the allowed ranges of the variables. However, there is reason to believe that the resulting production function could be used for short to medium-term forecasting purposes.

It is evident that (16.3) is essentially a scheme for approximating Solow's technical progress function in terms of the standard variables K, L, E. On reflection, there are many

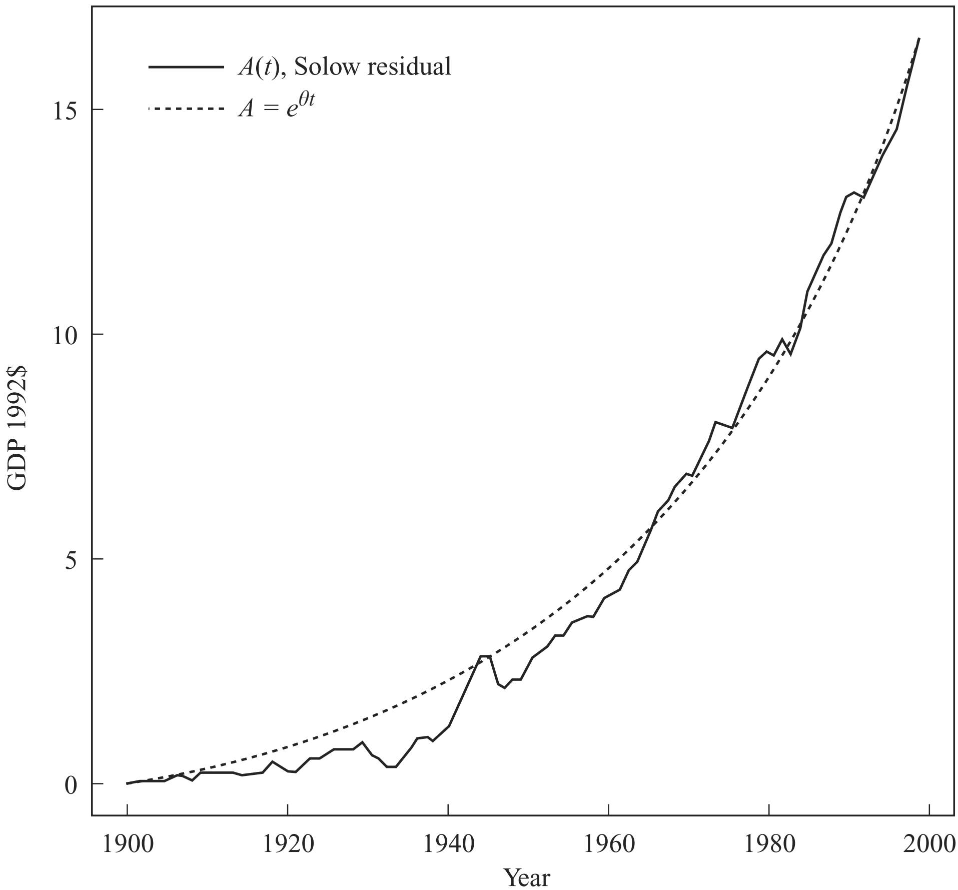


Figure 16.8 Technical progress function with best fit A*: USA, 1900–1998*

possible functional combinations that may satisfy the requirements. Indeed, other variables may actually serve the purpose better than the choice *E*. In particular, the extent of electrification of the economy has considerable intuitive appeal as a direct measure of technical progress.

This would greatly simplify the data problems, especially for application over a wide range of countries. Moreover, these preliminary results suggest directions for further research, leading, it is hoped, to useful policy implications.

17. Transmaterialization

Walter C. Labys*

Long-term materials demand patterns are important to examine because of the possibility of resource depletion as well as the long lead times required to create new mineral productive capacity. Since structural changes in materials demand are inevitably linked to the performance and adjustments of national economies, these changes have been historically measured relative to national income, employing a measure known as intensity of use (IOU). The demand declines observed in the IOU have been characterized as *dematerialization* or a decoupling of the materials sector from the industrial and other sectors of the economy. However, a preferable view is that the demand decline observed can be more aptly explained by *transmaterialization.* Transmaterialization implies a recurring industrial transformation in the way that economic societies use materials, a process that has occurred regularly or cyclically throughout history. Instead of a once-and-for-all decline in the intensity of use of certain materials, transmaterialization suggests that materials demand instead experiences phases in which old, lower-quality materials linked to mature industries undergo replacement by higher-quality or technologically more advanced materials.

The purpose of this chapter is to provide an explanation and evidence for transmaterialization. It consists of four parts: background, the dematerialization concept, the transmaterialization concept and empirical evidence.

BACKGROUND

The concept of dematerialization as developed in the 1980s can be said to be applicable only to a select group of technologically inferior materials, and not to an overall decline in the use of materials in general. Throughout history, the introduction, growth and decline of materials have been recorded as newer, more technically advanced materials have come into use. Several ages have even been named after the dominant materials consumed during their span as witness the 'Stone Age', the 'Bronze Age' or the 'Iron Age'. When we examine individual materials, boilers in the early 1800s were made of cast iron or sheet iron; by the 1860s, steel boilers were being used in response to the need for weight reductions in order to increase efficiency and to reduce costs. Materials used in the construction industry have gone through similar changes over time. Natural stone was probably the first mineral commodity used by modern man. Dimension stone has been used for several millennia as a construction material. Since the late 1800s, the use of dimension stone in building has been partially replaced by concrete, glass and bricks, because of the

* Thanks are due to Haixiao Huang for his technical assistance.

superiority of the latter materials in that they were stronger, less heavy and less costly. In roofing, clay and slate tiles have been replaced by sheet metal, wood shingles, asbestos–cement shingles and synthetic materials. In response to the need for more fuel-efficient automobiles, aluminum has significantly replaced steel in the manufacture of lighter-weight cars. While aluminum earlier experienced very high demand growth, the newer aluminum alloys are now being challenged by a new breed of materials, including advanced alloys, ceramics and composites (Eggert 1986).

THE DEMATERIALIZATION CONCEPT

A number of studies in the 1980s stressed the concept of dematerialization, that is the prospect that the USA and other national economies were experiencing a permanent decline in the use of materials in industrial production. In general, these studies have had three major limitations. First, they have taken a very short-run perspective, often including data only since 1970. Second, they typically cover only metals and industrial minerals. And third, few of them have included the 'life cycle theory' of product development in explaining the perceived changes in materials consumption. They thus ignored the possibility that, with changing needs, economies will replace old materials with newer, technologically more advanced materials in a cyclical fashion.

Much of this research began with Malenbaum's (1978) *World Demand for Raw Materials in 1985 and 2000*. That work also was one of the first to analyze materials demand employing the IOU method and surmised that an inverted U-shaped curve could be empirically observed from the IOU data, reflecting an initial rapid increase in the use of minerals as per capita GDP increases, then followed by a slow decline. Malenbaum, however, focused only on a small group of minerals while making many subjective judgments as to changes in IOU. In addition, he erroneously assumed that declining IOU occurred because of a shift in demand from manufacturing to the less materials-intensive service sector in the industrialized countries. It has been shown in other studies that employment has declined in the manufacturing sector, most likely because of increases in productivity, but that the demand for manufactured goods has not significantly declined relative to the service industries. Also the limitation of a small group of materials examined is that they were largely older minerals, neglecting composites, plastics and advanced ceramics.

A variation on Malenbaum's IOU methodology was utilized by Fischman (1980) in his *World Mineral Trends and US Supply Problems,* which found downturns in IOU for several of the seven metals analyzed (aluminum, chromium, cobalt, copper, manganese, lead and zinc) over the period from 1950 to 1977. Humphreys and Briggs (1983) examined the consumption trends for 12 metallic and 19 non-metallic minerals in the UK from 1945 to 1980. They found that the consumption of most minerals in the UK displayed a tendency to stagnate prior to the early 1970s, and that the consumption of non-metallics had shown a faster growth as compared to the metallics, indicating that their share of the total value of minerals consumed in the UK had increased significantly.

About the same time, Tilton (1985) in his study of 'Atrophy in metal demand' examined seven metals of which the growth in consumption had mostly declined since 1974.

Although Tilton implied that a structural transformation has been occurring in the US materials industries since the mid-1970s, the metals examined (aluminum, copper, steel, lead, tin, zinc and nickel) were, excluding aluminum, linked to mature basic industries. While the study found the IOU of each of the metals to be declining, each of these metals had been in use for more than 100 years and the total consumption of each of them had peaked decades ago. In addition, most of these metals had been or are being replaced by technologically more advanced and lighter high-performance metals. Other attempts to explain dematerialization can be found in a special metals demand conference proceedings by Vogely (1986), and international investigations were made by Lahoni and Tilton (1993) and by Roberts (1996). More recently, Humphreys (1994) and Moore *et al.* (1996) have examined changing IOU in the construction materials industry.

The main challenge to the materials intensity concept was made by Auty (1985) in his ‘Materials intensity of GDP’. Auty reviewed the above studies by Malenbaum and Fischman as well as by Leontief *et al.* (1983) and Radcliffe *et al.* (1981) to determine the reliability of their measures of declining materials IOU and to explain better their perceived trends. He disputed the inevitability of structural change in minerals for several reasons: substitution between materials tends to be erratic over time; the range of materials we use is widening rapidly as new technologies are employed, a fact that dematerialization does not take into account; and changes in the mix of manufacturing activity are proceeding faster than changes in the overall composition of GDP. He thus suggested that an alternative route to determining the direction of structural change and tracing underlying trends in minerals intensity could be provided by research on long wave economic cycles.

This was confirmed in works of Larson, Ross and Williams (1986) who provided evidence of some earlier or pre-World War II downturns in materials IOU and of Clark and Flemings (1986) who demonstrated that technological processes cause fluctuations in the way in which materials are used. The implications of these insights are that levels of IOU change regularly for different materials and that cyclical swings in this index might be a better indicator of mineral industry adjustments than that of a declining trend. This view was also supported by Sterman (1985) who concluded from his systems dynamics research and analyses of IOU patterns that structural changes in the economy can be better described as following a cyclical rather than a declining trend pattern. Finally, Ayres and Ayres (1996) show how dematerialization can be better explained in terms of materials substitution and recycling strategies.

THE TRANSMATERIALIZATION CONCEPT

This idea that materials undergo life cycles and substitution was furthered in the development of the new concept of transmaterialization; see Labys (1986), Labys and Waddell (1989), Waddell and Labys (1988) and Hurdelbrink (1991). Cyclical changes are in contrast to structural changes that imply growing obsolescence but not awareness of product life cycles. Transmaterialization describes the characteristic behavior of material markets over time by focusing on a series of natural replacement cycles in industrial development. As needs of economic society change, industries continually replace old materials with newer, technologically more advanced materials. This is part of the scientific process and, therefore, should not only be observable, but also be predictable from the point of view of profitability of individual mineral firms. Many developed countries have thus undergone an industrial transformation in which materials basic to 20th-century society are being replaced by materials with ramifications to the 21st century.

The origins of transmaterialization can be found in several aspects of the growth literature. Schumpeter (1927) developed a theory supporting the view that growth comes in spurts and appears as cyclical upswings. According to Schumpeter, progress is due to economically induced innovations, their gradual adoption and successful entrepreneurship. A more familiar notion of growth and one which underlies the Schumpeterian idea of progress specifies growth as following an S-shaped curve. Prescott (1922), Kuznets (1930) and Burns (1934) evaluated this growth theory for a sample of individual commodities and industries. Later Dean (1950) expanded this theory into the 'product life cycle' theory. The application of these theories to a number of different variables and different industries was later confirmed by Nakicenovic (1990).

The application of the life cycle model to transmaterialization requires five stages. The first model stage is the initial introduction of a new commodity. The performance of the material is not yet proven and sales are therefore sluggish. The consumption rates (measured as quantity/GDP) are typically low, along with vast potential markets. Representative of this stage are advanced ceramics, such as the silicon carbide and silicon nitride-based ceramics. These newer ceramics have been developed in order to fulfill a particular need for higher resistance to abrasion and to wear, high strength at high temperatures, superior mechanical properties, greater chemical resistivity and good electrical insulation characteristics.

The second, or growth, stage (sometimes referred to as the youthful stage) follows the discovery of a commodity or a major application. During this stage, consumption of the commodity increases rapidly as its properties are appreciated and promoted through research and dissemination of information. Consumption generally increases at a rate much faster than the economy as a whole, and this is reflected in a rise in the intensity of use index. Examples of youthful materials include gallium and the platinum group metals. During the third or mature stage, the growth in IOU begins to decline. The material has been accepted into industrial processes and the rapid growth of the youthful stage levels off. Aluminum represents a material currently in the mature stage.

According to Humphreys (1982), during the fourth or saturation stage, the IOU peaks and begins to decline, although the consumption, as measured in physical quantities, may still be increasing. Molybdenum, manganese and cobalt are currently in their saturation stage. The fifth or declining stage witnesses a significant decline in IOU of a material. During this stage, even total consumption declines, mainly because of newer materials replacement. Examples of materials in this last stage include tin, asbestos and cadmium.

Expanding upon the work of Humphreys and the life cycle theorists, Waddell and Labys (1988) showed that the recognition and the empirical determination of these cycles can make a strong case for transmaterialization. The hypothesis that growth and development occur in waves or cycles as defined in the product life cycle theory can be applied to materials markets. We would thus expect to see regular product life cycles for a number of minerals over extended periods of time. The timing and phases of these cycles obviously will vary with the nature of the products or minerals selected.

EMPIRICAL EVIDENCE

Labys and Waddell (1989) have provided empirical confirmation of the existence of these cycles and thus of transmaterialization for some 30 commodities. To provide a more aggregate demonstration of this phenomenon, these commodities have been further aggregated into five groups, each of which represents a different cyclical period in which intensity of use has peaked and declined or has increased. The grouping of the commodities and the periods they represent have been summarized in Table 17.1. The first group consists of those materials which experienced a peak in their IOU prior to World War II. Iron ore and copper are two of the materials included. The second group consists of those materials having their IOU peak just after World War II. Examples of materials in this category include nickel and molybdenum. The third group consists of materials for which the IOU peaked during the period from 1956 to 1970, namely manganese, chromium and vanadium. The fourth group consists of materials for which IOU peaked after 1970 and includes phosphate rock, aluminum and cobalt. The fifth and final group consists of materials for which the lOU has yet to peak. This group consists of newer, lighter, more technologically intensive materials, such as the platinum group metals, titanium, plastics and advanced ceramics (Mangin *et al.* 1995).

To provide further but brief evidence of the cyclical character of transmaterialization, the materials intensity of use (IOU) data have been updated from the original Labys and Waddell study. Sources for the commodity consumption data include the *Mineral Commodity Summaries* (originally the US Bureau of Mines but now the US Geological Survey) and for GDP the *Survey of Current Business* (US Bureau of Economic Analysis). The summary materials group indices have accordingly been updated and appear in Figure 17.1. Beginning with the Group Index 1, those materials appeared to have experienced rapid growth until the 1920s, followed by a phase of moderate growth lasting until the 1940s, when the IOU peaked. The phase of rapid growth of the materials found in Group Index 2 began in the late 1930s and lasted until after the end of World War II. Figure 17.1, suggests that growth then continued at a moderate rate and peaked soon thereafter, decline beginning around 1955. The upswing of the materials Group Index 3, which includes the years 1934 to the mid-1950s, increases until 1957, with a definite decline beginning in the early 1960s. The consumption of the materials contained in Group Index 4 continues to increase, but at a decreasing rate, so that the IOU is declining. The growth in IOU began in the late 1940s, peaked in the early 1980s, and is now in a declining phase. The Group Index 5 features those materials currently in their rapid growth stage. This phase of their life cycle began in the 1970s and has not yet peaked.

In conclusion, the examination of longer periods of changing intensity of use suggests that dematerialization might reflect short-term changes in materials consumption patterns. But, over the longer run, the transmaterialization concept provides a more realistic view of the way changes in materials consumption are likely to occur.

Table 17.1 US materials groupings, end uses and periods of peak intensity of use

Materials groups	Materials included	Major end-use sector	Time span*	Peak of intensity of use
1	arsenic	glass manufacturing	1885–45	1935
		industrial chemicals	(1941)	
	copper	electrical equipment		1943
	iron ore	construction industry		1941
		transport industry		
	lead	electric industry		1941
		chemical industry		
	tin	containers		1930
	zinc	iron and steel industry		1941
		construction		
2	asbestos	friction products	1925–70	1949
		insulation	(1949)	
	bismuth	pharmaceuticals		1948
		industrial chemicals		
	molybdenum	steel industry		1955
	nickel	iron and steel industry		1941/66
3	manganese	steel industry		1955
	vanadium	iron and steel industry	(1956)	1942/66/80
		construction		
	chromium	stainless steel		1957
	lithium	nickel and iron alloys		1955
		manufacture of aluminum		
4	aluminum	electrical applications	1945–86	1972
		packing industry	(1973)	
		construction industry		
	cobalt	super alloys		1952/75
	barite	oil and gas industry		1956/80
	phosphate rock	agriculture (fertilizers)		1979
	rutile	pigment		1974
5	gallium	electronics	1955–present	1979+
	geranium	electronics	(climbing)	1985+
	hafnium	nuclear reactors		1982+
	platinum metals	automotive industries		1983+
		chemical industries		
	titanium metals	aerospace industry		1969/80+
	rare earth elements and yttrium	catalysts		1985+
		electronics		
	polyethylene	packaging		1985+
	ceramics	optical fibers		
		machine parts		
		magnet components		
	composites	aerospace and automotive		

Note: *Peak use for group indicated in parentheses.

Source: Walter C. Labys, and L.M. Waddell (1989), 'Commodity life cycles in US materials demand', *Resources Policy*, **15**(3), 238–51.

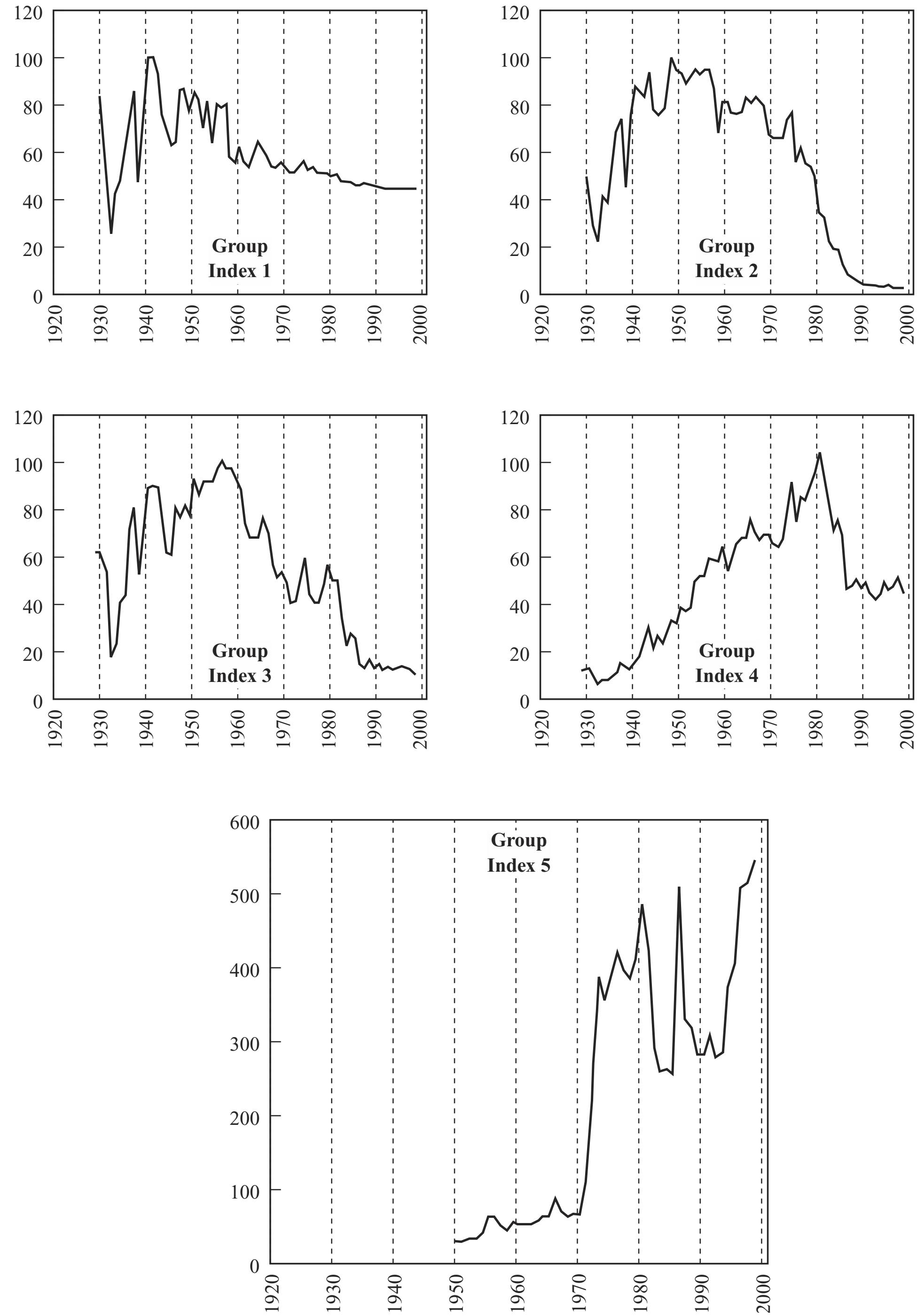


Figure 17.1 Materials group indices of intensity of use

18. Dematerialization and rematerialization as two recurring phenomena of industrial ecology

Sander De Bruyn*

Consumption of materials and energy is an important interface between the economy and the environment. Analyses of the patterns, causes and effects of materials and energy consumption are therefore very relevant in industrial ecology. The concept of dematerialization refers to the decline of material use per unit of service output. Dematerialization can be an important factor in making industrial societies environmentally sustainable: first, because dematerialization contributes to relieving scarcity constraints to economic development, and second, because, *ceteris paribus*, dematerialization reduces waste and pollution since, owing to the law of conservation of mass, every material resource input sooner or later turns up as emission or waste. However, dematerialization does not necessarily mean that wastes are minimized and material cycles are closed. Dematerialization is therefore equivalent to lowering the level of industrial metabolism without ensuring that the metabolism moves towards the more nearly closed cycles that can be found in ecosystems. (In the heart of the industrial ecology movement in the USA, the emphasis has traditionally been more on re-use, recycling and materials cascading than on dematerialization; see, for example, Frosch and Gallopoulos 1989.)

Several historical investigations suggest that dematerialization is occurring spontaneously in some developed countries (see, for example, Larson *et al.* 1986; Jänicke *et al.* 1989; Nilsson 1993) and in the material content of individual products, such as automobiles (see, for example, Herman *et al.* 1989; Eggert 1990). Labys and Waddell (1989) offered a different perspective, noting that much of what appears to be dematerialization may better be interpreted as transmaterialization (a shift from one group of materials to another). See Chapter 17.

Some authors have claimed a large technological potential for future dematerialization (for example, von Weizsäcker *et al.* 1997). One thesis that has been put forward is that in the process of economic growth the economy 'delinks' itself from its resource base, so that rising per capita income levels become associated with declining consumption of resources and their associated pollution (Malenbaum 1978; World Bank 1992b). But Auty (1985) has remarked that the reasons behind the dematerialization phenomenon are poorly understood. The fundamental questions are: is dematerialization really an aspect

* This chapter is taken in part from Chapter 8 in *Economic Growth and the Environment: An Empirical Analysis* by Sander M. de Bruyn (2000) and Chapter 10 in *Managing a Material World: Perspectives in Industrial Ecology*, Pier Vellinga, Frans Berkhout and J. Gupta (eds) (1998). Wherever I came across new data, insights or references, I have included them here.

of economic development, and can historically observed trends be extrapolated in the future?

This chapter aims to summarize some recent empirical findings in relation to resource consumption in developed economies. Surprisingly, the empirical evidence hints at the possibility that dematerialization is only a temporary phenomenon and could be followed by a period of rematerialization. There are four sections: a historical overview of resource consumption facts as presented in the literature since the beginning of the 1960s; a discussion of the idea that a period of dematerialization may be followed by one of rematerialization; explanations for the observed patterns of dematerialization and rematerialization; conclusions.

HISTORICAL TRENDS

Until the late 1960s, the consumption of materials, energy and natural resources was widely assumed to grow at the rate of economic growth. This gave rise to growing concerns about Earth's natural resource availability, notably in the Club of Rome's report on 'Limits to Growth' (Meadows *et al.* 1972). This report modeled a linear, deterministic relationship between economic output and material input. As a consequence of worldwide economic growth, mankind would face widespread resource exhaustion, which in turn would negatively affect economic and population growth, human health and welfare.

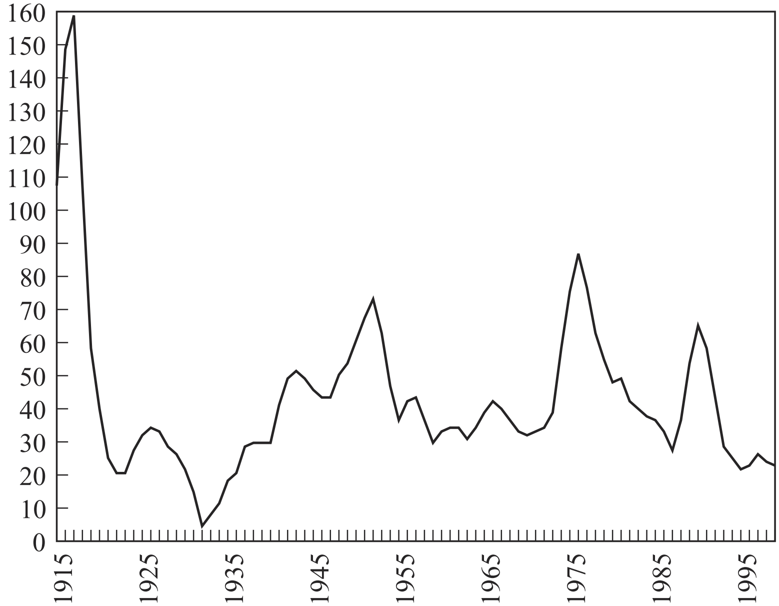
The arguments put forward by Meadows *et al.* were a modern restatement of much older views originating with Malthus and Ricardo. They predicted that scarcity of natural resources (including land) would eventually result in diminishing social returns to economic efforts, thus limiting the possibility of increasing economic welfare in the face of population growth. The best end result would be a steady state, with a constant population, bounded by the carrying capacity of the earth.

The notion of scarcity was critically examined by Barnett and Morse. They state:

> Advances in fundamental science have made it possible to take advantage of the uniformity of energy/matter, a uniformity that makes it feasible, without preassignable limit, to escape the quantitative constraints imposed by the character of the earth's crust. A limit may exist, but it can be neither defined nor specified in economic terms. Nature imposes particular scarcities, not an inescapable general scarcity. (Barnett and Morse 1963, p. 11)

In effect, they suggest that progression in human knowledge opens up new substitution possibilities and advances the technology of extraction, use and recycling, all of which prevents resource scarcity from becoming a constraint to economic activities. Simon (1981) has in this respect referred to human knowledge as 'the ultimate resource'.

Barnett and Morse (1963) postulated that growing scarcity would necessarily be reflected in higher prices for resources, yet their study revealed no indication of rising prices for mineral resources in the USA since the mid-19th century. The price of zinc in the USA relative to the consumer price index (the price of all other consumer goods) provides an example (Figure 18.1). Despite wartime fluctuations, prices remained relatively stable, showing no sign of growing scarcity. This can be explained economically in terms of the effect of price mechanisms on resource markets. The price for a (mineral) resource is determined by four interrelated factors: (a) demand for the mineral; (b) supply available from known reserves (those deemed profitable to develop); (c) supply from recycling; and (d) supply of, and demand for, substitutes. Even under conditions of fixed technology, price increases tend to be compensated by falling demand and increasing supply as more reserves (both of virgin and of recycled materials) become economically exploitable. The development of technology for exploration and extraction is also stimulated by price increases, actual or anticipated. All these factors have tended to keep resource prices declining over the long run.



Source: *Mineral Factbook*; American labor statistics.

Figure 18.1 *Three-year moving averages of prices of zinc relative to the consumer price index in the USA (1975 = 100)*

Several critics have pointed out that prices give limited information on scarcity per se because they reflect other information as well. Mineral markets are traditionally oligopolistic. Observed price declines might, in some cases, be the result of more intense competition, reducing monopolistic profits. Another possibility is that primary producers may have over-estimated future economic growth and thus over-investing in production capacity. Finally, market prices do not reflect important externalities, such as social costs of mine and smelting waste disposal (and pollution associated with energy use) which are not paid by the miner and are therefore not included in the price of the mineral (Cleveland 1991). For all these reasons, prices may be poor indicators of scarcity, especially when interpreted in the context of environmental impacts.

The changing pattern of materials demand in itself also seems to deny the limits-to-growth predictions. In the 1980s, some economists discovered a decline in materials demand growth, which was later interpreted as dematerialization. Table 18.1 shows that, between 1951 and 1969, the consumption of most refined metals increased exponentially. Annual growth rates were often higher than 5 per cent, meaning that a doubling of metal consumption would occur every 15 years. Predictions on future demand for materials by Landsberg *et al.* (1963), US Bureau of Mines (1970), Meadows *et al.* (1972) and Malenbaum (1978) depicted declining, but still rather high, growth rates for the next decades. But if we compare these predictions with the actual developments of the world materials demand we see what statisticians would call 'a break in series'. World growth rates of metals between 1973 and 1988 have approximated a modest 1 per cent per annum which implies that consumption will double only every 70 years.

Table 18.1 Annual world growth rates in the consumption of refined metals

	Actual[a] 1951–69	Meadows[b] 1971–	Malenbaum[c] 1975–85	Actual[d] 1973–88
Iron ore	6.2	1.8	3.0	0.8
Copper	4.7	4.6	2.9	1.2
Aluminum	9.2	6.4	4.2	1.7
Zinc	4.9	2.9	3.3	0.7
Tin	1.7	1.1	2.1	−0.5
Nickel	5.0	3.4	3.1	1.7
GDP[e]	4.8	NA	4.5	3.0

Sources: (a) Tilton (1990a); (b) Meadows *et al.* (1972) – no end year given in this estimate; (c) Malenbaum (1978); (d) World Resources Institute (1990); (e) *UN Statistical Yearbook*, various issues.

Explanations for the slackening of materials demand were formulated by Malenbaum (1978) as the 'intensity of use hypothesis'. In brief, the demand for materials is derived from the demand for final goods: from housing and automobiles to beer cans. Because raw material costs form only a small proportion of finished product cost, they have an insignificant influence on demand. Instead, income is the explanatory factor in materials consumption.

Malenbaum (1978) predicted non-uniform income elasticities over time and across countries because of the different characteristics of the composition of final demand associated with different stages of economic development. Developing countries with an economic structure relying on subsistence farming typically have a low level of materials and energy consumption. But as industrialization increases, countries specialize first in heavy industries to satisfy consumer demand for durables, plus investment demand for infrastructure. During this phase materials consumption increases at a faster rate than income. The subsequent induced shift towards service sectors results in a decline in the materials intensity of current demand.

Thus Malenbaum depicted the relationship between materials demand and income as an inverted U-shaped curve (*IUS* in Figure 18.2 with the turning point at ***a***). It should be noted that Malenbaum has presented his theory, not for the absolute consumption of materials but for the relative consumption of materials: the amount of materials per unit of income. This is the 'intensity of use', which would follow a similar inverted-U curve but with a lower turning point (in Figure 18.2, ***b*** would be the turning point of this curve).

This has no implications for the elaboration of the theory presented here because Malenbaum acknowledged that further movements along the inverted-U curve would eventually result in absolute reductions of materials consumption.

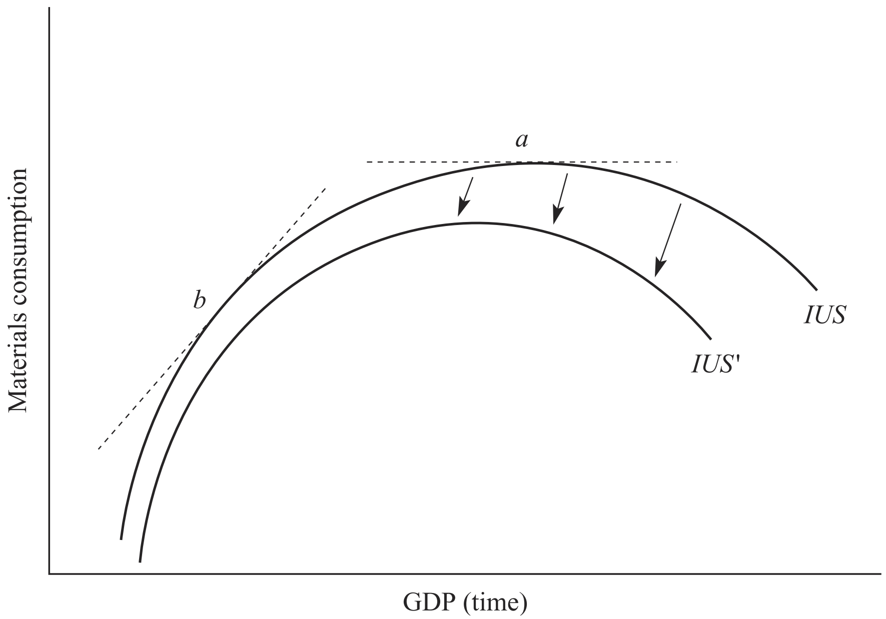


Figure 18.2 The 'intensity of use' hypothesis and the influence of technological change

Technological change has the effect of shifting the relationship between materials demand and income downwards. The same economic value can be generated with less material input because of technological improvements in materials processing, product design and product development. Late developing countries follow a less materials-intensive development trajectory. The implication of Malenbaum's theory is that, in the long run, the growth in world materials consumption levels off and eventually starts to decline. This last stage could be labeled 'strong dematerialization', implying an absolute decrease in the consumption of materials (de Bruyn and Opschoor 1997).

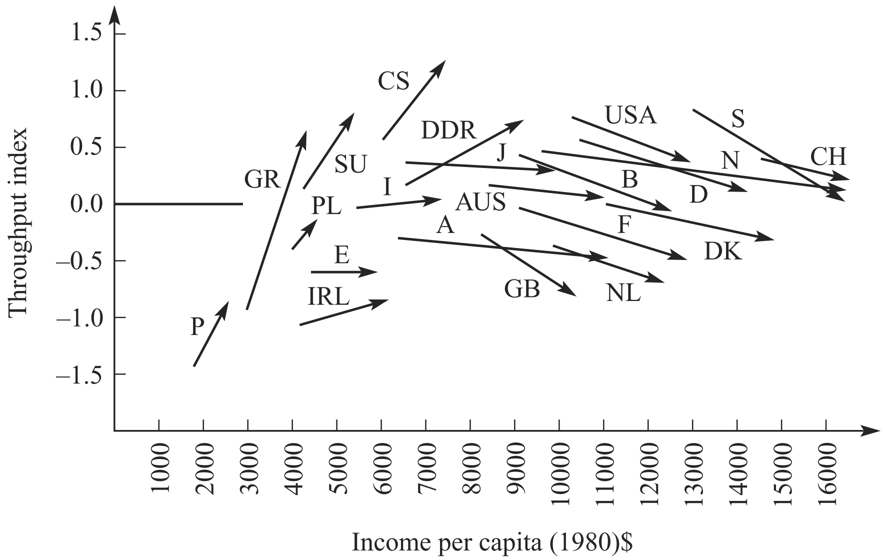
WILL DEMATERIALIZATION CONTINUE?

The 'intensity of use' hypothesis has found support in a number of case studies on the consumption of some specific materials and energy (for example, Bossanyi 1979; Chesshire 1986; Williams *et al.* 1987; Tilton 1986, 1990; Valdes 1990; Goldemberg 1992; Nilsson 1993). These show dematerialization occurring in a wide range of developed countries since the early 1970s, often 'strong dematerialization'. However, Labys and Waddell (1989) have emphasized that conclusions based on studies that take only a few materials into account may be misleading. Comparing the trends in consumption of some 30 materials in the US economy, they conclude that the phenomenon may sometimes be more adequately described as 'transmaterialization', or substitution between groups of materials, for example plastics for metals.

The environmental implications are not neutral. Resource consumption has consequences for the environment by virtue of the mass balance principle (Ayres and Kneese 1969). There would be no reason to assume that environmental pressure decreases owing to dematerialization if only the composition of the materials and energy consumed changed but not the total quantities. In fact, new substances may enter the environment with more serious negative impacts than the ones they replace. For example, the impacts of DDT, CFCs and PCBs on human health and the environment were only proved long after their market introduction.

The possibility of transmaterialization implies that more comprehensive indicators for material consumption should be used when patterns in materials and energy consumption are interpreted in the light of overall environmental pressure. Such indicators can be seen as representative of the total flow of 'throughput', defined by Daly (1991b, p. 36) as the physical flow of matter/energy from the environment, through the economy and ultimately back to nature's sinks. Only a few empirical studies have formulated and analyzed such indicators over time. Ayres (1989a) presents data for the US economy for four years between 1960 and 1975. Bringezu *et al.* (1994) estimate total material consumption (including removal of earth for mining) for West Germany for five years between 1980 and 1989 (see Chapter 23). More recently, a consortium of institutions (Adriaanse *et al.* 1997) has investigated the resource inputs of 30 substances for four countries (the USA, Japan, the Netherlands and West Germany). All of these studies aggregate throughput on the basis of mass (weight) and focus on materials flows as inputs to an economy. The results of these studies do not confirm the hypothesis that strong delinking between total aggregated material throughput and economic growth has already occurred. In all cases, aggregated material consumption has been increasing. Berkhout (1998) concludes that the resource consumption per unit of GDP has been falling, however.

This is in contrast to the conclusions of a study by Jänicke *et al.* (1989) which used an indicator approximating the volume of throughput (materials and energy), calculated with a simple statistical aggregation technique. This throughput indicator consisted of the consumption of steel, the consumption of energy, the production of cement and the volume of transported materials and products by rail and road. The authors argued that cement and steel were chosen because these involve highly polluting processes and to a large extent they involve the physical realities of industrial end products, construction and machinery industry (ibid., p. 173). Comparing 1970 and 1985 for a set of 31 OECD and centrally planned economies, they concluded that economic growth is already delinked from the throughput indicator for the more developed economies. The results are summarized in Figure 18.3 where the arrows give the linearized developments in the throughput indicator between 1970 and 1985 for various countries. The pattern seems to confirm the earlier analysis by Malenbaum: rising levels of throughput in less developed economies and decreasing levels of throughput for the more prosperous countries. This figure suggests that the phenomenon of strong dematerialization also holds for a more comprehensive set of matter/energy flows and it has been interpreted by some commentators as 'a sign of hope' in resolving environmental problems (cf. Wieringa *et al.* 1991; Simonis 1994; von Weizsäcker and Schmidt-Bleek 1994). A more detailed description of throughput indicators used in some studies can be found in de Bruyn (2000).



Note: Arrows indicate the linearized development between 1970 and 1985.

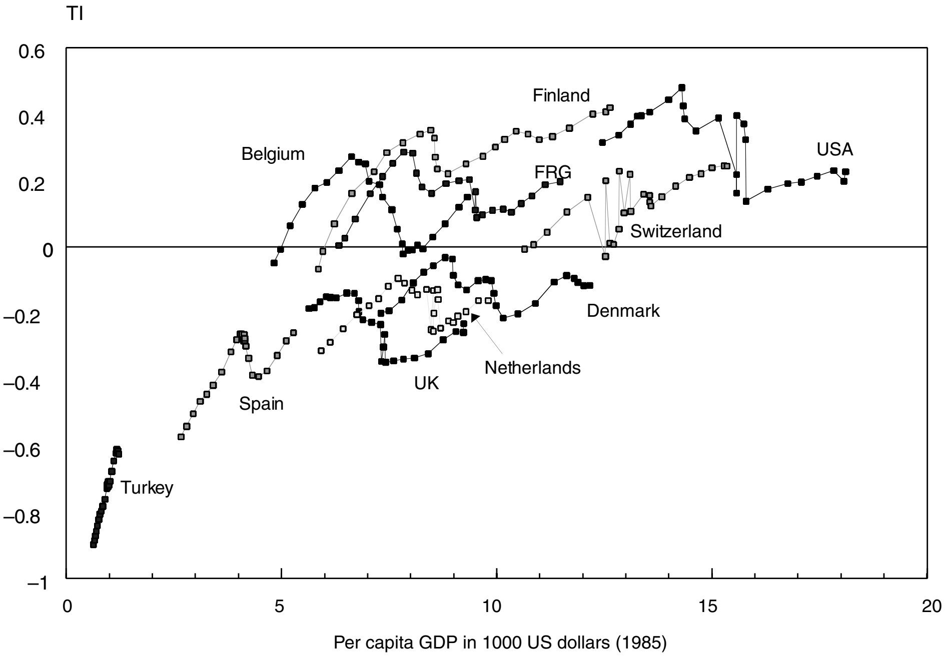
Source: von Weiszäcker and Schmidt-Bleek (1994), after Jänicke *et al.* (1989).

Figure 18.3 Developments in aggregated throughput

The results of Jänicke *et al.* were re-examined by de Bruyn and Opschoor (1997), extending the time-horizon and making some minor improvements in the indicator calculation. Our results suggest that, since 1985, there has been an upswing in the levels of throughput for several developed economies. Figure 18.4 makes this development explicit for eight countries between 1966 and 1990. Using the same indicators as Jänicke *et al.* (1989) we see that the developed economies experienced an increase in their levels of throughput again after 1985. Between 1973 and 1985 a dematerialization tendency was observed in all countries except Turkey. But during the late 1980s many countries showed increases in throughput almost equivalent to the growth in GDP. Hence it seems that the actual pattern of throughput over time may be more adequately described as 'N-shaped', similar to the inverted-U shaped curve but with a subsequent phase of 'rematerialization'. Similar evidence of a recent phase of rematerialization has been provided recently by Jänicke (1998) and Ko *et al.* (1998). The question is whether rematerialization is an anomaly or part of a broader relationship between resource consumption and economic growth. I consider this next.

AN EVOLUTIONARY PERSPECTIVE ON DEMATERIALIZATION

Dematerialization can be the result of technological and structural changes in the use of materials (Malenbaum 1978). Technological changes imply increases in the efficiency of



Note: Every dot indicates the moving average over three years between 1966 and 1990.

Figure 18.4 Developments in the throughput index (TI)

material use through, for example, improved processes or product design. Structural changes can be defined as those changes in the composition of economic activities that have an impact on resource use. Three types of structural changes are normally mixed in the literature. They refer to (a) a change in the structure of inputs, that is, a shift in the relative shares of capital, labor and various types of natural resources in production processes; (b) a change in the structure of production, that is, a shift in the relative shares of various sectors that make up the economy; and (c) a change in the structure of consumption, that is, a shift in the composition of consumption due to changes in life styles. Both structural and technological changes can be influenced by a mix of variables, such as resource prices, governmental policies, consumer preferences and so on.

The phenomenon of dematerialization has often been explained by reference to structural changes. The intuitively appealing notion is that citizens in developing countries first show an appetite for material welfare (cars, infrastructure, consumer durables) which increases total material consumption, and that only at certain high income levels do services (banking, insurance, education, entertainment) become more important. Structural changes thus provide a logical explanation for an inverted U-shaped pattern of resource use. However, they do not explain an N-shaped pattern. The idea that consumers, in the course of economic development, start to prefer material consumption goods again, after a period in which they preferred more services, is untenable from a theoretical perspective as well as from an intuitive point of view.

However, empirical support for structural changes has been meager and unconvincing. There exists some empirical work decomposing the change in energy intensities into structural and technological factors. Howarth *et al.* (1991) and Binder (1993) have decomposed the change in energy intensities for a range of OECD economies. They find little support for structural changes as an important determinant of the recorded decreases in the energy intensities between 1973 and the early 1990s. The decreases in energy intensities are much better explained by referring to technological improvements in processes and product innovations. The absence of structural changes in materials demand can also be explained by reference to rebound effects. As early as 1864 the economist Jevons remarked that the savings in coal for steam engines due to technological progress are not effectuated owing to the growing demand for transport (Ko *et al.* 1998). Also Herman *et al.* (1989) have suggested that dematerialization in production may not be realized because of growing resource use in consumption. If consumers benefit financially from savings in material use in the production stage, they may spend their additional income on new consumer goods, so that the total effect can be negative. This effect was also recently demonstrated empirically by Vringer and Blok (2000).

This evidence points to the notion that patterns of resource consumption hinge critically on the development of technology over time. In economics, two conflicting views on the development of technology exist. In neoclassical economic theory, technological change follows a process of Darwinian natural selection at the margin. Whereas technological change was first assumed to be 'autonomous' and 'exogenous' to the neoclassical model, the theory of endogenous growth has more recently incorporated technological change by explicitly investigating the role of human knowledge in generating R&D and welfare. Romer (1990), for example, argues that economic growth can be enhanced by investing in 'human capital' that results in innovations and technological change. Whether innovations are rejected or accepted depends on the chances of the firm to compete more successfully in the market. Technological change is thus endogenized by making it dependent on a cost–benefit analysis concerning investments. The yields of those investments gradually improve over time because of the accumulation of knowledge. A logical implication is that the economy gradually becomes less material-intensive and more knowledge-intensive.

Alternatively, it has been suggested that the process of technological change does not follow a smooth process along a path of equilibrium, but is characterized by disequilibrium *and* an evolutionary path of learning and selection (Dosi and Orsenigo 1988). Innovations over time may typically come in certain clusters as the result of a process of 'creative destruction' (Schumpeter 1934). This view is supported by the accumulating evidence of biological evolution as a 'punctuated equilibrium'. Fossil evidence suggests that species remain virtually unchanged for quite a long time, but unexpected quantum leaps result in sudden appearances and extinctions. Evolution occurs not so much on an individual level but more on a macro level, and species evolve together in their environment. Gowdy (1994) applied these findings to economics and proposed that the economic system may be relatively stable and in equilibrium during a certain period of time, followed by drastic shifts (or shocks) in technological paradigms and institutional and organizational structures.

These two approaches imply a different pattern of throughput over time. The first approach seems to be compatible with the inverted-U curve in which marginal changes allow for gradually falling intensities over time. The second approach, however, is capable of explaining the N-shaped pattern in which the (temporary) dematerialization phase may be the result of a drastic shift (or shock) in technological and institutional structures. In this view the equilibrium stage may be represented by a positive linkage between throughput and income. As the result of a shock, the relationship is reversed and a period of dematerialization follows. After the effects of the shock have evaporated, the relationship returns to its equilibrium stage and rising incomes result again in rising throughput.

These two conflicting views can be investigated empirically by mapping the intensity of use in a phase diagram showing the dynamics over time. Ormerod (1994) did this for employment and found patterns of relative tranquillity in employment/GDP ratios, followed by periods of high volatility. He emphasized that various attractor points can be found in the data; the level of unemployment/GDP hovers at a certain level for a long period of time, then suddenly starts to drift until a new equilibrium level is reached. This led him to conclude that the neoclassical assumption of gradual change is not supported by the data.

Evidence for the existence of attractor points in the relationship between certain types of mass/energy throughput and income can be found in Figures 18.5 to 18.8. These figures give the development of the consumption of energy and steel per unit of GDP in the UK and the Netherlands for data in the period 1960–97 (data sources: IEA 1995; Eurostat various). The intensity of use is here plotted against two dimensions: the value in the current year and the value in the previous year. All figures show evidence of attractor points in the data. Figure 18.5 shows that, in the UK, apparent steel consumption per unit

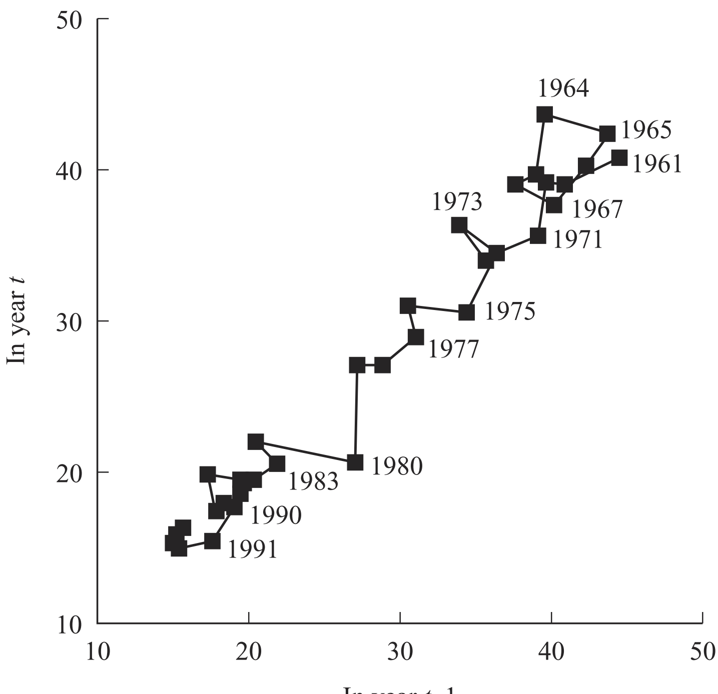


Figure 18.5 Steel intensities in the UK, 1960–95, in kg/1000 US$ (1990)

of GDP remained relatively stable in the 1960s, moving around an attractor point of 42g/US$ (in price levels of 1990). However, after 1970 and especially after 1974, steel intensities started to fall, continuing until 1983 when a new attractor point was reached around 20g/US$. This attractor point remained stable until 1990, after which the intensities started to fall again and stabilized at a level of 16g/US$.

A similar pattern can be found with reference to the energy intensities (Figure 18.6). One attractor point existed during the 1960s at a level of 0.33 ton oil equivalent (toe) per 1000 US dollars. After 1973, intensities started to fall continuing at least until 1988, when a new attractor point was reached at 0.22 toe/1000$. The steel intensities in the Netherlands follow a similar pattern, with two distinct attractor points: the first lasted from 1960 to 1973 and the second from 1982 to 1995 (Figure 18.7). Energy intensities in the Netherlands (Figure 18.8) show three distinct attractor points: one in the 1970s, circulating around a level of 0.3 toe/1000$, and second and third attractor points from 1983 to 1988 and 1990 to 1996. More evidence for attractor points in the consumption of steel and energy in the USA and West Germany can be found in de Bruyn, 2000.

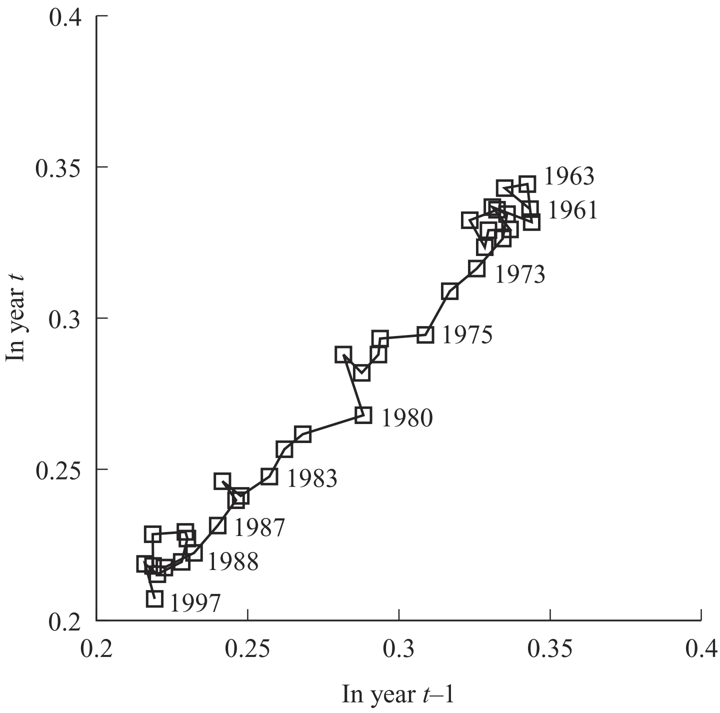


Figure 18.6 Energy intensities in the UK, 1960–97, in toe/1000 US$ (1990)

These patterns suggest that the theory of punctuated equilibria may more adequately describe the patterns of material and energy consumption over time than theories based on marginal and gradual change. The evidence cited above suggests that, when the economy is in an equilibrium phase, the intensities of materials and energy remain constant and move slightly around a certain attractor point. These marginal fluctuations

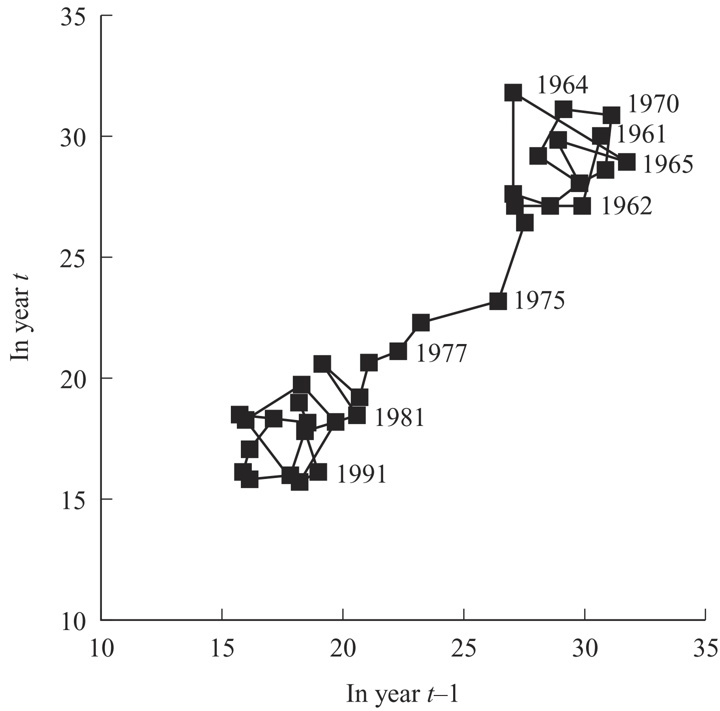


Figure 18.7 Steel intensities in the Netherlands, 1960–95, in kg/1000 US$ (1990)

can be described as 'business cycles'. The intensities of use remain relatively stable and economic growth has the effect of an equiproportional increase in the consumption of materials and energy. Hence the equilibrium state of the economy implies that materials consumption and economic growth are perfectly linked.

However, during times of (radical) shifts in technological and institutional paradigms, intensities start to fall rapidly and throughput declines, at least until the economy stabilizes again around a new attractor point. Then the positive relationship between economic growth and materials consumption is restored and throughput rises again at approximately the same rate as the growth in incomes until a new technological or institutional breakthrough enables another dematerialization phase. The result is an N-shaped relationship between income and throughput in the medium term, as discussed in the previous section. In the long run a saw-like pattern will appear, in which periods of dematerialization are followed by periods of rematerialization. The effects on long-term total throughput cannot be stated beforehand. It depends on the duration of the rematerialization and dematerialization phases. However, the data in Figure 18.4 suggest that rematerialization may be as important globally as 'strong dematerialization' since higher-income countries also have higher levels of throughput.

This pattern of resource use also clarifies why it has been so difficult in the past to estimate the future development of material use (see Table 18.1). The econometric implication is that there is no stable relationship between material use and income. In the absence of a stable relationship between materials demand and income, predictions cannot be

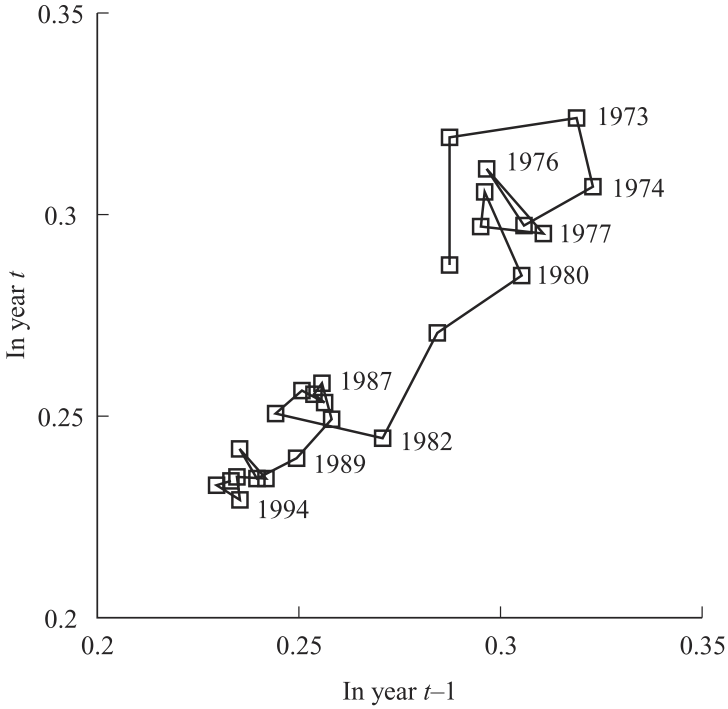


Figure 18.8 Energy intensities in the Netherlands, 1970–96, in toe/1000 US$ (1990)

made if the nature of the shocks is not understood. (A shock can be defined as an influence on one variable in an estimated relationship which is not explained by the other variables. As a shock is exogenous to a certain specified model, inclusion of other variables may endogenize such a shock and this certainly will improve predictions. Imagine, however, how difficult it must have been to predict the oil shock back in 1971.) This point was discussed by Labson and Crompton (1993) but has remained largely ignored in the literature, probably because it deals with more sophisticated econometric techniques such as cointegration.

DISCUSSION AND CONCLUSIONS

It has been suggested that the relationship between throughput and income has an inverted U-shape in the sense that rising incomes can be associated with lower levels of resource inputs in developed economies. This chapter has discussed and reanalyzed this evidence and finds no support for strong dematerialization. Empirical work discussed here suggests that, in the medium term, the relationship between income and throughput is probably N-shaped, with a saw-tooth pattern in the very long term. An explanation for this pattern is that, in an equilibrium phase, economic growth results in an equiproportionate increase in throughput. However, during times of radical changes in the technological and institutional paradigms, the relationship between throughput and income growth may be altered somewhat discontinuously, owing to technological advances in the processing and use (including substitutions) of materials.

Such a phase of dematerialization will not continue indefinitely, and the positive relationship between income growth and throughput growth is likely to be restored, albeit at a lower level of throughput per unit of GDP. The empirical data presented here also show where comparisons of data on throughput between the beginning of the 1970s and the mid-1980s go wrong: these years belong to two different classes of attractor points. One cannot conclude that dematerialization is occurring in developed economies from comparisons such as those conducted by Jänicke *et al.* (1989). Investigations into the patterns over a longer period of time are required to determine the true relationship between resource use and income. Predictions on the future development of material use and dematerialization must take into account the stochastic imbalances in the relationship between material use and income.

The analysis conducted here is not limited to material and energy use. Some empirical work has suggested that there exists an inverted-U pattern, or environmental Kuznets curve, between several pollutants and income (cf. Selden and Song 1994; Grossman and Krueger 1995). Given the fact that emissions and wastes originate from the consumption of materials and energy, it can be expected that some pollutants would follow similar patterns and that in the long run the relationship between emissions and income might be N-shaped. Some evidence for this has recently been gathered by de Bruyn (2000) and Stern and Common (2000) who conclude that, owing to stochastic imbalances, the environmental Kuznets curve is likely to be spurious.

With worldwide continuing economic growth and developed countries being currently in an equilibrium phase of throughput and income, issues of scarcity, availability, exhaustion of natural resources and growing pollution may return to the research and political agendas. This is likely to occur after the current relinking phase has ended and prices of resources start to rise once again. (In fact, petroleum prices started rising rapidly in 1999, and the higher levels show every indication of being permanent.) Institutional and technological breakthroughs will then be required to reverse the current rematerialization phase.

One such period of radical change obviously occurred in the years following the first oil crisis (1974–5) when prices of energy and raw materials rose to unprecedented levels and environmental awareness was increasing. This may have prompted governments and business enterprises to reconsider their use of resources and to start a process of rationalization, or restructuring. A revival of environmental awareness may be the vehicle through which a new stage of restructuring may be introduced so that the positive relationship between income growth and throughput growth will again be shifted in a different direction, albeit only temporarily.

19. Optimal resource extraction

Matthias Ruth

THE ECONOMICS OF RESOURCE USE

Economics has traditionally concerned itself with 'the best use of scarce means for given ends' (Robbins 1932). Typically, the 'ends' are considered to be achieved when consumers realize maximum possible satisfaction – 'maximum utility' – from the consumption of goods and services. The economy helps realize maximum utility in a three-step procedure. First, resources are extracted and transformed into goods and services. Second, producers supply goods and services to consumers for final use. Finally, wastes from production and consumption are recycled in, or removed from, the economic system.

Typically, market mechanisms are assumed to reconcile the independent decisions of producers and consumers, and to result in a final coherent state of balance between demand for goods and services and supply. This state of balance for an economy is known as a general economic equilibrium (Walras 1954 [1874], 1969). Given the knowledge about consumers' preferences, resource endowments, technologies and market forms, economists can compute the prices and quantities of goods and services that are consistent with economic equilibrium. The equilibrium state can then be used as a reference against which to compare the impacts of alternative actions, such as government interventions in markets, on prices and quantities of goods and services, and the subsequent welfare effects for the economy (Arrow and Hahn 1971).

Instead of dealing with the simultaneous choice of all producers and consumers in an economy, this chapter concentrates on economic models dealing with the optimal extraction of natural resources. The models are, therefore, by their nature partial equilibrium models – all economic conditions outside the resource-extracting sector are assumed to be given, and *within that context* an equilibrium for extracting firms is identified. The next section addresses basic concepts in economics that are used to model and inform decision making, such as the optimal extraction of resources. The subsequent section then presents a basic theoretical framework that is common to most economic models of resource extraction. That section is followed by a discussion of challenges to the theory and application of optimal resource extraction models. The chapter closes with a review of topics of current research in the economics of resource extraction.

ASSUMPTIONS ABOUT DECISION MAKING

Four concepts of economics play a crucial role in describing and guiding economic decisions: opportunity costs, marginalism, substitution and time preference (Ruth 1993).

Opportunity costs are defined as utility or profits forgone upon choosing one situation or action over another. Thus the concept of opportunity costs enables economists to rank alternative choices. In a dynamic context, opportunity costs not only are defined by the utility or profits forgone in the period in which a decision is made but depend on the effects of this decision on all future alternatives. Application of the opportunity cost concept forces decision makers to compare the cost and benefits of a wide range of alternative actions against each other. And since in many cases gradual changes in behavior are possible, economists are interested in the extent to which, for example, a small additional amount of a resource extracted today rather than tomorrow influences overall utility or profits.

The use of marginal analysis helps identify implications of small, continuous adjustments in behavior, and is inherently linked to the calculus of variations. If economic relationships between, for example, consumption and utility, production inputs and outputs, and extractive effort and yield are expressed in the form of continuous functions, the impacts of *marginal* changes of resource extraction, production or consumption on utility or profit can be readily calculated.

Any particular consumption and output levels may be achieved by substituting alternative actions or goods. Such a substitution is assumed to be possible at least at the margin, that is, within a relevant range of alternatives, small changes can be made by slightly altering single alternatives and combining alternatives to enlarge the set of choices. In the case of resource extraction, when faced with ore of lower grades or resource deposits at greater depths, a firm may respond by increasing its inputs of labor or capital to counteract declining physical resource accessibility.

Since economic activities such as production and consumption take place over time, these activities must be ranked according to their occurrence in time. Consumers and producers must choose actions that maximize utility or profits not only within a period of time but over a set of periods extending into the future. Time preference is expressed by decision makers taking into consideration the temporal distribution of consequences resulting from their actions. For example, present consumption is typically preferred to future consumption, thereby reflecting the time preference of economic agents.

Aggregation of utility or profits that occur over time is achieved by discounting future utility. The rate at which future utility is discounted is determined by the time preference of the consumer and may capture a variety of factors such as elements of risk and uncertainty associated with a consumption plan.

The discount rate reflects the trade-off between current and future consumption that consumers are willing to accept. Analogous to discounting future utility, producers discount profits to compare alternative production and investment decisions that affect profits over multiple periods of time. One risk in choosing a specific extraction plan may be associated with the advent of substitutes or new technologies that reduce the economy's need for the resource. Typically, it is argued (Solow 1974a) that the discount rate represents an allowance for the ability of future generations to make better use of the remaining resource stock than the present generation. Thus both the time preference and the rate of productivity of natural and human-made capital must be reflected in the discount rate.

A host of practical issues still apply when deciding which discount rate to use. For practical purposes, a market rate of interest for capital is typically used to reflect the opportunity cost of investment, and the associated risks and uncertainties (Lind *et al.* 1982).

BASIC MODELING FRAMEWORK

At the center of traditional resource economics is the analysis of alternative plans for the extraction of natural resources. Optimality is achieved by maximizing the present value of utility, profits or welfare from resource extraction. This maximization is done under a set of constraints that describe the resource base, the firms' technological and organizational structure, consumer behavior and market characteristics. In a general equilibrium model additional factors, such as competition between virgin and recycled materials, would also be taken into account.

Non-renewable Resources

Gray (1913, 1914) was the first to recognize that the conditions for optimal resource depletion are different from the optimality conditions for the production of ordinary goods (Fisher 1981). A basic assumption is that a non-renewable resource can be extracted and used only once. Therefore optimal prices of a unit of a resources must reflect not only its cost of extracting an additional unit but also account for the opportunity costs associated with depleting the resource endowment by that unit.

In his path-breaking article, Hotelling (1931) formalized the conditions for the optimal extraction of a non-renewable resource through time. In the basic setting, identical resource owners compete with each other on perfectly competitive resource markets with perfect information. Perfect competition implies that the extraction quantity by each firm is small in comparison with total demand for the resource so that no owner can individually affect the resource's market price. However, price changes in response to the collective, yet independent, actions of the resource-extracting firms. By assumption, all resource-extracting firms also face perfect competition for labor and capital inputs into their operation, no substitutes exist for the resource, all initial resource endowments are known, no replenishment of the resource stock is possible and no new deposits are found. The response of demand for the resource to changes in extraction quantities is given and known to all participants on the resource market. Each of the resource-extracting firms knows how their costs of extraction change as extraction quantities change, and all are aware of the opportunity cost of operating their resource-extracting enterprise.

The assumed goal of the resource-extracting firms is to maximize – under the given conditions – the discounted value of all profits that can be made from the date at which extraction begins to the terminal date at which resource endowments are depleted. The key question to be answered by the decision makers then is: how much of the resource should be extracted each period?

A decision maker may choose not to extract any of the resource early on, and to wait until the competitors run out of the resource. At that point prices will be high and large profits may be reaped. However, since that situation may occur in the distant future, the discounted value of those future profits will be low. Alternatively, a decision maker could extract all of the resource early. Since costs are high at high rates of extraction, and since the price of the resource is low in periods of plentiful supply, lower profits can be expected. Yet those profits may be invested to earn interest for the firm.

An optimal extraction path is characterized not by either extreme of extracting all today or all in the future, but by a point of indifference between extracting an additional unit today and leaving it in the ground for extraction in the future. Marginal analysis guides the choice of optimal extraction quantities. The discount rate reflects the corresponding trade-off between these decisions.

In his seminal paper, Hotelling (1931) showed that the optimum strategy for resource-extracting firms is to choose extraction quantities such that the price of an exhaustible resource minus marginal extraction costs rises over time at exactly the rate of discount, which is generally equated with the interest rate. This difference between price and extraction cost is referred to as the 'scarcity rent' of a resource. Since Hotelling's time, many of his restrictive assumptions have been relaxed. For instance, modern resource extraction models allow for imperfect competition or imperfect information.

Renewable Resources

Renewable resources exhibit regeneration and thereby introduce a set of dynamics and complexities into economic models of optimal resource extraction that are not encountered in models of non-renewable resources. In the case of renewable resources it is possible to have steady-state optimal solutions with non-zero resource use.

Models for optimal use of renewable resources differ largely according to the type of resource analyzed, for example, the biological characteristics of the resource. Basic versions of economic models of harvesting trees and catching fish are described in this section, and the general conclusions drawn here are broadly applicable to many other renewable resources.

One basic model of optimal harvest of trees assumes that discounted utility from total harvest from all age classes be maximized (Clark 1976). Each age class is assumed to consist of trees of the same age which increase their biomass at a prescribed rate of growth. Harvesting implies clearcutting even-aged stands with trees of uniform size and leaving the cleared area for natural regeneration. The objective is to identify the optimal rotation period. It can be shown that for the steady-state optimum the percentage growth rate of trees must equal the percentage value of an infinite stream of marginal changes in asset values (Faustmann 1849; Hellsten 1988; Wear and Parks 1994). Thus only two factors influence optimal harvest times: the growth rate of plants and the discount rate.

The basic model of renewable resource extraction is the starting point of a variety of studies that are used to ascertain the optimal rotation period for various management objectives (Chang 1984) and evaluate effects of changes of the economic assumptions on optimal management plans (see Reed 1986 for a review). For example, in order to enhance applicability, the basic model was extended to include stand volume as an independent variable, thereby allowing for the simultaneous determination of optimal thinning of stands and rotation ages (Cawrse *et al.* 1984) and to include predicted future stand density for the whole stand as a function of current stand age and density (Knoebel *et al.* 1986).

Models of optimal catch, for example of fish, follow a logic similar to that of models of optimal forestry – stock sizes are specified by initial conditions and growth rates, and 'harvest' rates are chosen such that the cumulative discounted value of profits is maximized. However, a carrying capacity is typically defined to specify changes in growth rates with changes in stock size (Conrad and Clark 1989). In an unregulated fishery, fishermen will be attracted to the fishery until there is no net flow of revenue from fishing. The resulting equilibrium stock size is a function of the unit cost of effort, price of landed fish, fishing efficiency and the discount (interest) rate. That equilibrium tends to be quite fragile (Myers *et al*. 1997; Ruth and Hannon 1997). Decreases in cost of catch (resulting from technological improvements), or increases in landing price, efficiency or interest rates may result in economically optimal effort levels that drive resource stocks to zero. As a consequence, a 'potentially' renewable resource can be exhausted.

CHALLENGES TO THEORY AND APPLICATION OF OPTIMAL RESOURCE EXTRACTION MODELS

The previous section introduced the basic framework for the determination of optimal resource extraction from non-renewable and (potentially) renewable resources. This section turns to three major issues surrounding the formulation and application of models of optimal resource extraction: discounting, empirical applicability and the limits of partial equilibrium analysis of resource use. The following section then closes with a brief summary and an overview of the role that models of optimal resource extraction may play in the future.

Discounting

The question is often raised whether society should discount future utility or profits at all. Discounting implies that we value consumption and production in future periods less than present consumption and production. Thus, for decisions with ramifications that extend over long time frames, discounting implies that we value the needs of future generations differently from those of present generations, possibly leading to rapid resource exhaustion. Social discounting is typically justified by the assumption that technological improvement will automatically give rise to increasing economic wealth, and that even though future generations inherit smaller *biophysical* resource endowments, an enlarged stock of human-made resources will compensate for the reduction in the physical resource base. However, the assumption can be questioned. The *possibility* that future generations will be better off than the present generation is not a certainty.

There is another problem surrounding the use of a discount rate besides the ethically controversial issue of treating different generations differently. This problem is due to differences in social and individual discount rates. Discount rates applied by individual consumers or producers do not correspond necessarily to discount rates that may be applied by society as a whole to evaluate economic actions, such as the extraction or harvest of natural resources. This issue has caused considerable discussion in the economics literature (Lind *et al.* 1982; Portney and Weyant 1999). The choice of the discount rate is vital to the evaluation of economic activities. The discount rate determines whether an action has positive present value of profits or utility, whether it is better than others (has higher present value of profits or utility in the set of possible actions) and whether its timing is optimal (for example, whether waiting would resolve uncertainty and, thus, lead to higher present value of profits or utility).

Once a discount rate is chosen for the evaluation of alternative consumption and production plans, the question is whether this rate can be assumed to remain constant over time. Discounting at a constant rate seems appropriate if economic agents assume that the probability of factors affecting the choice among actions remains constant over time (Heal 1986). Since the determination of a social discount rate is already controversial (Lind *et al.* 1982; Portney and Weyant 1999), assumptions about time dependence of the discount rate are not likely to be accepted easily.

The choice of discount rate reflects time preference and changes in an economy's productivity that result from converting biophysical resources into reproducible, human-made capital. Not all of the energy and material resources, however, are used to provide goods and services for consumers or to produce new capital equipment. Some resources are used to produce and store the information that describes production processes. One may put a value on such accumulated knowledge in analyzing the economics of materials recycling and energy resource depletion, focusing on ways to preserve economic efficiency while addressing the issue of intergenerational equity (Page 1977; Ruth and Bullard 1993). Page proposed a 'conservation criterion' to ensure an intertemporally egalitarian distribution of exhaustible resources. This conservation criterion states that each generation that irreversibly depletes energy resources or highly concentrated ores has an ethical obligation to leave the next generation enough new technology to produce the same utility from the more dilute resources. He cites the example of improved mining technology or the discovery of new reserves to ensure the next generation's access to the same quantity of low-cost reserves as the present generation inherited. The value of technology, in Page's framework, would be indicated by the level of severance taxes on energy or other resources needed to stimulate the development of such knowledge endowment for the next generation.

Empirical Relevance

Although Hotelling's basic approach is widely accepted among resource economists, empirical evidence for the Hotelling Rule on the basis of a firm's behavior is weak and, with few exceptions (Stollery 1983; Miller and Upton 1985), rather disappointing (Smith 1981; Farrow 1985). The lack of empirical support for the Hotelling Rule is partly due to the fact that the Hotelling's model does not explicitly take into account a firm's production capacities, capital requirements, capital utilization and time adjustments in production technologies.

Traditional models of optimal non-renewable and renewable resource extraction are also simplistic with respect to the behavioral assumptions on which they are built. For instance, short-run and long-run decisions are typically not distinguished (Bradley 1985). Decision makers may not attempt to identify optimal extraction paths over, potentially, many decades to centuries, but rather choose two or more sets of time periods over which decisions are made. As a consequence, one discount rate may be applied over decisions in the immediate future, while different discount rates may be chosen for the medium to long term.

Hotelling-style models on a macroeconomic level are more abundant than their microeconomic counterparts. With these models, time paths of various scarcity measures are investigated on an economy-wide basis. Potential candidates for economic measures of scarcity include resources prices, marginal extraction cost or scarcity rent rates (Brown and Field 1979; Hall and Hall 1984).

One of the most influential studies is that of Barnett and Morse (1963) for mineral resource depletion in the USA during the time from the civil war to 1957. In their analysis, Barnett and Morse define increasing resource scarcity by increasing real unit cost of extractive products. The hypotheses of an increase in real unit cost of extractive products and an increase of real unit cost of producing non-extractive commodities are rejected. On the basis of these findings, Barnett and Morse (1963) argue that, with the exception of forestry resources, extractive resources in the USA did not become more scarce during the time span considered. Their findings are confirmed by Barnett (1979) but rejected by Smith (1979a) who both update Barnett and Morse's study.

Though these studies are questioned frequently as to their methodological deficiencies, they initiated discussion about both the measurement of resource scarcity and the adequacy of various economic scarcity measures (see Brown and Field 1979; Fisher 1979; Smith 1980; Hall and Hall 1984; Farrow and Krautkraemer 1991; Norgaard 1990) as well as the empirical evidence of non-increasing resource scarcity. General agreement has not been achieved yet and is likely not to occur as long as measures for resource scarcity are tied to economic performance only and do not account for the underlying biophysical reality of economic activities and interactions of economic performance and environmental quality with material cycles and energy flows through the entire ecosystem (Ruth 1993).

Slade (1982, 1985) incorporated exogenous technical progress and endogenous change in ore quality into an optimal control model of resource depletion. A U-shaped trend for resource prices is shown to give a better fit to historic data than linear price trends. Thus she concludes that long-term price movements tend to exhibit upward shifts in resource prices in response to increasing scarcity while technical progress allows for only intermediate price decrease. Slade's analyses redirected the discussion about resource scarcity towards the importance of technical change and exogenous effects on resource depletion (Mueller and Gorin 1985).

Perhaps the most devastating critique of the use of Hotelling-style models for the empirical assessment of resource scarcity has been voiced by Norgaard (1990). He contends that efforts to detect resources scarcity on the basis of optimal resource extraction models fall victim to the following logical fallacy. Optimal extraction models are based on the premises that if (a) resources are scarce, and (b) resource-extracting firms know about the scarcity, then economic indicators reflect scarcity. Empirical studies attempt to track changes in scarcity indicators through time. If, for example, unit cost of extraction, resource prices or scarcity rent rates rise, the conclusion is drawn that resources have indeed become scarce. However, to logically conclude resource scarcity from observed changes in scarcity indicators assumes that premise (b) is fulfilled: that is, that those making the decision about extraction quantities know about the extent of resource's availability. However, if resource-extracting firms already know about the scarcity of the resource, then the exercise of detecting resource scarcity from the empirical record is moot. It would be simpler to ask the decision makers in resource-extracting firms directly.

Models of renewable resource extraction are prone to much of the same critique as models of non-renewable resources. The applicability of a variety of approaches to modeling optimal harvest rates for renewable resources is discussed in considerable detail in Getz and Haight (1989). The models and methods described there do considerably more justice to the biological aspects of growing and harvesting natural resources than the basic model described above. However, common to all those models is the fact that optimal behavior is guided by growth rates and the discount rate. When growth rates are exogenous to the model, the discount rate is the sole determinant of optimal harvest and, as a consequence, the representation of feedbacks between the ecosystem that supports resource growth and economic activity is severely limited.

Partial Equilibrium Approach

By its nature, economic theory is anthropocentric and, thus, selective in the consideration of effects of economic actions on the environment and the role of environmental goods and services for economic activities. It is consumer utility, welfare or profit that is maximized under a set of constraints that are given by the environment and recognized by economic decision makers. Such constraints reflect, for example, the finiteness of an essential resource or the growth rates of plants harvested or animals caught. However, many other important environmental constraints are typically not captured fully in the economic decision process. Rather, these constraints are captured only as far as they impose apparent, immediate restriction on the deployment of the economically valued factors of production. A variety of constraints that are associated with unpriced material and energy flows that may lead to fundamental changes in the physical or biotic environment are frequently not (but can, in principle, be) considered. An obvious example is climate change which is being induced, in part, by the emissions of greenhouse gases from combustion of fossil fuels, methane and carbon emissions resulting from land use changes, and reduced carbon sequestration from biomass harvesting.

Economic actions, such as the extraction or harvest of a resource and the production of goods and services, are accompanied by changes in the state of the economic system and its environment. Production of goods and services in the economy necessitates use of some materials and energy that are typically not valued economically. Additionally, production inevitably leads to waste of materials and energy, thereby affecting the long-term performance of the ecosystem. Models of optimal resource extraction are particularly selective in the consideration of such feedback processes between the economic system and the environment. This is not to say that economics altogether disregards them. All models and theories provide abstractions of real processes. However, neglecting some physical and biological foundations of economic processes may lead to results that neglect vital issues, such as the earth's capacity to support life, which ultimately determine economic welfare. The complex interdependencies between economic decisions and the degradation of the environment due to material and energy use are often neglected or treated as 'externalities'; that is, they are treated as effects that are not a priori part of the decision process but that can be considered a posteriori in economic decisions if there is economic value associated with them. The treatment of important interdependencies of the economic system and the environment as externalities, without restructuring the theory, amounts to making ad hoc corrections introduced as needed to save appearances, like the epicycles of Ptolemaic astronomy (Daly 1987, p. 84).

THE HISTORY AND POTENTIAL FUTURE OF MODELS OF OPTIMAL RESOURCE EXTRACTION

The economics of resource extraction is part of a larger body of theory that identifies, under a set of assumptions about resource endowments, preferences, technologies and market forms, the conditions that meet economic criteria such as maximal profits, minimal cost or largest economic welfare. The key concepts that help determine such conditions and guide economic decision making are the concepts of opportunity cost, marginalism, time preference and substitution. The concepts of opportunity cost and marginalism are useful in everyday decision making: decision makers must compare a wide range of alternative actions and carefully fine-tune their decisions. The concept of time preference reflects asymmetries in choices today and in the future. And the concept of substitution reflects the fact that there are different ways of using resources to achieve desired ends.

The emerging debate about sustainable use of resources forces decision makers to broaden their view and to compare alternative actions with respect to their long-term and system-wide costs and benefits. Over larger scales, preferences and technologies vary, in part owing to the diversity of producers and consumers. Environmental conditions are far from stable, and may themselves exhibit complex (even chaotic) behavior. Markets do not exist for many of the relevant flows of materials and energy between the economy and its environment, and within the environment. The laws of thermodynamics – especially the concepts of mass and energy balances and of exergy – can be used to trace changes in the quantity and quality of materials and energy within and across economic and environmental systems. By keeping track of mass and energy flows, and the biophysical processes that they trigger in the environment, a basis can be created on which to judge alternative production and consumption before mechanisms get established to internalize externalities. Resource extraction may then be optimal not only with respect to the narrowly defined partial equilibrium criteria of traditional economic models of resource use, but also with respect to issues concerning the long-term and large-scale performance of an economy that is an integral part of a changing ecosystem.

Models of optimal resource use may use insights from industrial ecology to help identify sustainable resource use – as opposed to the more narrowly defined optimal extraction of natural resources – in four major ways. First, the use of physically based measures of material and energy flows within an economy and between the economy and its environment provide information that can be used to minimize harm to humans and their environment. Second, that information may be used to identify new business opportunities. Third, being able to properly address diversity and complexity of economy–environment interactions in economic models is central to an understanding of what it means to become sustainable, and how to design policies and institutions that help society achieve sustainability. Fourth, by enriching existing methodologies with engineering realism and environmental realism, economic modeling may better contribute to management and policy decision making.

20. Industrial ecology and technology policy: Japanese experience

Chihiro Watanabe

Despite many handicaps, Japan achieved extraordinarily rapid economic development over the four decades preceding the 1990s. This success can be attributed, in part, to technology as a substitute for constrained production factors such as energy and environmental capacity. While technology played a significant role in driving a positive (feedback) cycle of economic growth, its governing factors interrelate with each other as in a metabolic system. Consequently, during the 'bubble economy' in the latter half of the 1980s and its implosion in the early 1990s, Japanese industry experienced a structural stagnation in R&D activities. This, in turn, has broken the above virtuous cycle, and growth has stalled.

The global environmental consequences of environmental emissions from fossil energy use are causing mounting concern regarding the long-term sustainability of our industrial system. The necessary response to this concern is to find a solution which can overcome energy and environmental constraints without destroying the drivers of growth. An approach to such a solution can be regarded as a dynamic game involving the 'three Es': economy, energy and environment. For simplicity, these can be represented as aggregate production (Y), energy consumption (E) and carbon emissions (C), the latter being a surrogate for all emissions associated with carbon-based energy use. Economic growth can be represented by the identity

$$\Delta Y/Y = \Delta C/C - \Delta\,(E/Y)/(E/Y) - \Delta\,(C/E)/(C/E). \tag{20.1}$$

Options for sustainable growth can be characterized in terms of the following variables: carbon emissions (C), energy efficiency (E/Y) and decarbonization or fuel switching (C/E). Table 20.1 compares the development paths of Japan, the USA, Western Europe, the former USSR and Eastern Europe, and LDCs for the 10 years following the second energy crisis in 1979 (1979–88). From the table we note that Japan recorded the highest average economic growth in that decade, 3.97 per cent per annum. Such growth was possible as a result of annual energy efficiency improvement of 3.44 per cent, a 0.59 per cent rise in fuel switching and a 0.06 per cent decline in CO_2 emissions. The less developed countries (LDCs) followed Japan in terms of GDP growth with an average annual growth rate of 3.53 per cent. During the 10-year period, fuel switching had a positive effect as it rose by 0.16 per cent. However, energy efficiency fell by 0.85 per cent per annum, leading to a 4.22 per cent annual increase in carbon emissions. The USA attained 2.78 per cent average annual GDP growth supported by a 2.62 per cent energy efficiency improvement and a 0.11 per cent rise in fuel switching. Carbon emissions increased by 0.05 per cent. In

Table 20.1 Comparison of paths in attaining development in major countries/regions in the world, 1979–88 (average change rate: % per annum)

	Production ($\Delta Y/Y$)	Energy efficiency ($\Delta(E/Y)/(E/Y)$)	Fuel switching ($\Delta(C/E)/(C/E)$)	CO_2 emissions ($\Delta C/C$)
Japan	3.97	−3.44	−0.59	−0.06
USA	2.78	−2.62	−0.11	0.05
W. Europe	2.01	−1.78	−1.33	−1.10
USSR/E. Europe	1.72	0.45	−0.83	1.34
LDCs	3.53	0.85	−0.16	4.22

Note:
a. Production is represented by GDP.
b. $\Delta Y/Y = \Delta C/C - \Delta(E/Y)/(E/Y) - \Delta(C/E)/(C/E)$Å@Å@ where $\Delta Y = dY/dt$.

Sources: Y. Ogawa (1991) using IEA's statistics, energy balances of OECD countries and energy statistics and balances of non-OECD countries.

Western Europe, GDP growth measured 2.01 per cent per annum as energy efficiency improved by 1.78 per cent, fuel switching increased by 1.33 per cent and carbon emissions decreased by 1.10 per cent. Average annual GDP growth in the countries of the former USSR and Eastern Europe was 1.72 per cent. Energy efficiency declined by 0.45 per cent while fuel switching rose 0.83 per cent. Carbon emissions increased by 1.34 per cent annually.

The relative advantages and disadvantages of energy efficiency improvement and fuel switching are generally governed by economic, industrial, geographical, social and cultural conditions. Japan's improvement in energy efficiency was initiated by industry as part of its survival strategy, to cut the cost of energy and, especially, imported petroleum. However, owing to lack of access to natural gas, Japan's fuel switching ability was limited. This was not the case in Western Europe, where nations were able to rely on natural gas from the North Sea. However, in contrast to Japan's example, the efforts of industry in Western Europe to increase energy efficiency were not strong.

Thus Japan's success in overcoming energy and environmental constraints while maintaining economic growth can largely be attributed to intensive efforts to improve energy efficiency. Technology played a key role through a combination of industry efforts and government policy, coordinated by the Ministry of International Trade and Industry (MITI) (Watanabe and Honda 1991). However, since the relaxation of energy constraints (starting in 1983), the sharp appreciation of the yen triggered by the Plaza Agreement (in 1985), the era of the 'bubble economy' (1987–90) and its collapse (1991), Japan's progress in substituting technology for energy has weakened significantly, leading to concerns about the future.

To date, a number of studies have identified the sources supporting Japanese industry's technological advancement (for example, the US Department of Commerce 1990; Mowery and Rosenberg 1989, pp. 219–37). Mansfield (1983) noted that federally supported R&D expenditures substituted for private expenditures. He concluded that, while the direct returns from federally financed R&D projects might be lower, the projects seemed to expand the opportunities faced by firms and induced additional R&D investments by them. Scott (1983) demonstrated Mansfield's postulate by providing supportive results such as the fact that government-supported R&D encourages company-financed R&D. The author identified similar functions in MITI's industrial technology policy (for example, Watanabe and Clark 1991; Watanabe and Honda 1991, 1992; Watanabe 1999).

A number of studies have attempted to quantify the substitutability of energy with other production factors (for example, Christensen *et al.* 1973; National Institute for Research Advancement of Japan 1983). However, most of these works deal with labor and capital (and energy); a few also deal with materials as a production factor. None has taken the technology factor explicitly into account. Although some pioneering work attempted to use a time trend or dummy variable as a proxy for technological change, such methodologies are hardly satisfactory for analyzing the non-linear effects of R&D investment. Watanabe (1992a, 1995a, 1995c) measured the stock of technological knowledge and incorporated it into a trans-log cost function. He was able to explain Japan's success in overcoming the effects of the two energy crises in the 1970s by investing in energy efficiency. Attempts have also been made to apply this mechanism to the global environment (Watanabe 1993, 1995b). This work suggests that the current stagnation in industry R&D might weaken the existing substitution, leading to the rise of energy (and environmental) constraints (Watanabe 1992b, 1995d). Given the comprehensive and systematic nature of the global warming and policy relevance to this issue, a comprehensive systems approach is essential.

The next section reviews MITI's efforts to induce energy efficiency R&D in industry. The following section introduces a quantified model to explain Japan's success in substituting technology for energy after the energy crises of 1974–9; it also discusses some limits of technology policy. The final section summarizes implications for sustainable long-term economic development.

MITI'S EFFORTS TO INDUCE ENERGY EFFICIENCY R&D

The model variables and data construction used in this section are as follows (see also Watanabe 1992a).

1. Production and production factors
   Y (production) = (gross cost at 1985 fixed prices),
   L (labor) = (number of employed persons) × (working hours),
   K (capital) = (capital stock) × (operating rate),
   M (materials: intermediate inputs except energy) = (intermediate inputs at 1985 fixed prices) − (gross energy cost at 1985 fixed prices),
   E (energy) = (final energy consumption),
   T (technology, as cumulative R&D investment, depreciated to reflect obsolescence).
2. Technology-related production factors
   Lr (labor for technology) = (number of researchers) × (working hours),
   Kr (capital stock of R&D: KR) × (operating rate),
   $KRt = \mathrm{GTCkt} + (1 - \rho\mathrm{kr})\mathrm{KRt} - 1$,
   $GTCk$ (R&D expenditure for capital at 1985 fixed prices),

ρ*kr* (rate of obsolescence of capital stock for R&D: inverse of the average of lifetime of tangible fixed assets for R&D),
Mr (materials for R&D),
Er (energy for R&D).

3. Cost
   GC (gross cost),
   GLC (gross labor cost) = (income of employed persons) + (income of unincorporated enterprises),
   GCC (gross capital cost) = (gross domestic product) − (gross labor cost),
   GMC (gross materials cost) = (intermediate input) − (gross energy cost)
   GEC (gross energy cost) = (expenditure for fuel and electricity),
   GTC (gross technology cost) = (R&D expenditure and payment for technology imports).
4. Technology-related cost
   GTCl (R&D expenditure for labor),
   GTCk (R&D expenditure for capital),
   GTCm (R&D expenditure for materials),
   GTCe (R&D expenditure for energy).

Table 20.2 compares trends in the ratio of government energy R&D expenditure and GDP in the G7 nations after the first energy crisis. Looking at the table we note that Japan, unlike other advanced countries, maintained a higher level of government energy R&D expenditure even after the downward movement in international oil prices (starting from 1983).

Table 20.2 Trends in the ratio of government energy R&D expenditure and GDP in G7 countries, 1975–94 – percentile (1/100%)

	1975	1980	1985	1990	1994
Japan	0.67	1.26 (0.32)	1.18 (0.26)	0.87 (0.16)	0.91 (0.23)
USA	0.77	1.46 (0.89)	0.60 (0.30)	0.45 (0.30)	0.33 (0.27)
Germany	1.18	1.28 (0.42)	0.93 (0.23)	0.35 (0.14)	0.19 (0.08)
UK	1.04	0.98 (0.29)	0.82 (0.26)	0.30 (0.10)	0.11 (0.07)
Canada		0.91 (0.51)	1.04 (0.62)	0.57 (0.32)	0.42 (0.20)
Italy		0.73 (0.09)	1.29 (0.10)	0.61 (0.40)	0.34 (0.18)
France				0.49 (0.10)	0.42 (0.20)

Note: Figures for Germany before 1990 are only for the FRG; figures in parentheses indicate the ratio of non-nuclear energy R&D expenditure.

Sources: *Energy Research, Development and Demonstration in the IEA Countries* (IEA, 1980), *Review of National Programmes* (IEA, 1981), *Energy Policies and Programmes of IEA Countries, 1987 Review* (IEA, 1988), *Energy Policies and Programmes of IEA Countries, 1994 Review* (IEA, 1995).

Table 20.3 summarizes trends in Japanese government energy R&D expenditure and MITI's share in 1980, 1985 and 1994. It indicates that MITI was primarily responsible for Japanese government non-nuclear energy R&D, with more than 90 per cent of the total government budget.

Table 20.3 Trends in Japanese government energy R&D expenditure and MITI's share (billion yen at 1985 fixed prices)

	Energy R&D total		Non-nuclear energy R&D	
	Gov. total	MITI (share)	Gov. total [Non-nuclear share]	MITI (share) [Non-nuclear share]
1980	310.1	81.3 (26.2%)	78.4 [25.3%]	71.2 (90.8%) [87.5%]
1985	371.6	115.1 (31.0%)	91.1 [24.5%]	89.7 (98.4%) [78.0%]
1994	403.0	112.6 (27.9%)	101.9 [25.3%]	92.2 (90.5%) [81.9%]

Source: *Energy Policies and Programmes of IEA Countries: 1996 Review* (IEA, 1997).

The government support ratio for energy R&D initiated by manufacturing industry was higher than 14 per cent over all of the periods examined, and it increased to more than 20 per cent after the first energy crisis in 1973. The corresponding ratio for non-energy R&D decreased as Japan's technological level improved, and it now stands at almost 2 per cent. These trends demonstrate the importance of Japan's energy R&D in terms of national security (Industrial Technology Council of MITI 1992).

The proportion of MITI's energy R&D budget to its total R&D budget increased dramatically, from 20 per cent before the first energy crisis to 35 per cent in 1975, 48 per cent in 1980, 65 per cent in 1984 and 43 per cent currently. The share of energy R&D expenditure in manufacturing industry reached its highest level in the early 1980s and then changed to a declining trend. It is currently 3.5 per cent, which is almost the same level as after the first energy crisis.

Japan has adopted different industrial policies throughout its economic development, all of which reflect the international, natural, social, cultural and historical environment of the postwar period. In the late 1940s and 1950s, the goal was to reconstruct its war-ravaged economy, laying the foundation for viable economic growth. During the 1960s, Japan actively sought to open its economy to foreign competition by liberalizing trade and the flow of international capital. In the process, supported by a cheap and stable energy supply, it achieved rapid economic growth (see Figure 20.1) led by the heavy engineering and chemical industries. Unfortunately, the concentration of highly material-intensive and energy-intensive industries led to serious environmental pollution problems, which necessitated a re-examination of industrial policy (Watanabe 1973; MITI 1993, pp. 276–307).

Recognizing the need for a change in direction, MITI formulated a new plan for Japan's industrial development. Published in May 1971 as 'MITI's Vision for the 1970s' (Industrial Structure Council of MITI 1971), this plan proposed a shift to a knowledge-intensive industrial structure which would reduce the burden on the environment by depending more on technology and less on energy and materials. The vision stressed the significant role of innovative R&D to lessen dependency on materials and energy in the process of production and consumption).

In order to identify the required basic policy elements to implement the vision, MITI organized an ecology research group in May 1971 (MITI 1972a). Consisting of experts

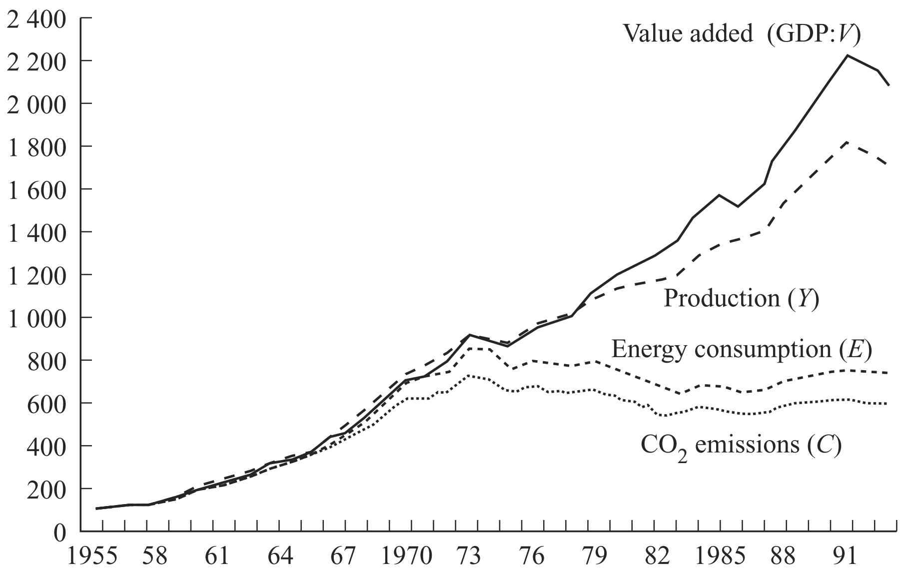


Figure 20.1 Trends in production, energy consumption and CO_2 discharge in the Japanese manufacturing industry, 1955–94 (Index: 1955 = 0.1)

from ecology-related disciplines, this group proposed the concept of 'Industry–Ecology' as a comprehensive method for analyzing and evaluating the complex mutual relations between human activities centering on industry and the surrounding environment (MITI 1972b). In the summer of 1973, MITI concentrated on developing R&D programs aimed at creating an environmentally friendly, yet efficient, energy system (MITI 1993).

The first energy crisis occurred a few months later. MITI focused its efforts on securing an energy supply in the face of increasing oil prices. Given these circumstances, it initiated a new policy based on the Basic Principle of Industry Ecology to increase energy security by means of R&D on new and clean energy technology. This policy led to the establishment in July 1974 of a new program, the Sunshine Project (R&D on New Energy Technology, MITI 1993). The Sunshine Project initiated this approach by enabling the substitution of technology for limited energy sources, such as oil. Further substitution efforts were to be made not only in the energy supply field but also in the field of energy consumption. Improvement in energy efficiency, through technological innovation, can cut dependence on energy sources. In line with this consideration, MITI initiated the Moonlight Project (R&D on Energy Conservation Technology) in 1978 (MITI 1993). The Sunshine Project and the Moonlight Project represented 4.9 per cent of MITI's total R&D budget in 1974, 13.8 per cent in 1979 and 28.9 per cent in 1982.

MITI's energy R&D policy during the 1974–87 period can be summarized as follows:

- encourage broad involvement of cross-sectoral industry in national R&D program projects such as the Sunshine Project and Moonlight Project;
- stimulate cross-sectoral technology spillover and inter-technology stimulation;
- induce vigorous industry activity in the broad area of energy R&D.

This inducement should then lead to an increase in industry's technology knowledge stock of energy R&D which has a transtechnological and sectoral stimulation function. This inducement can then play a catalytic role in industry's technology substitution for energy.

Coinciding with the establishment of the 'Sunshine Project' (1974) and the 'Moonlight Project' (1978), similar strategies for sophisticated energy use were postulated in the USA, including 'A Time to Choose' by the Ford Foundation (1974) and 'Soft Energy Paths' by Lovins (1977). The former stressed the significance of the redirection of federal energy R&D to goal-oriented programs with major goals of energy conservation, diversity of energy supplies and environmental protection. It argued that a major new thrust in R&D addressed to energy conservation opportunities was urgently needed. The latter was essentially an argument that nuclear power was going to be far more costly than its proponents had admitted. It did not provide a complete blueprint for an alternative R&D program. Neither of these (or other) major studies in the USA – mainly carried out by non-governmental organizations – succeeded in overcoming the bias of US industry towards increasing domestic energy supply rather than reducing demand.

TECHNOLOGY AS A SUBSTITUTE FOR ENERGY: AN EXPLANATORY MODEL

As already noted, Japan realized a notable improvement in its energy efficiency after the energy crises of the 1970s and was able to continue rapid economic development with a minimum increase in energy dependency and carbon emissions (Figure 20.1). In order to elucidate the sources of this dramatic trend shift, Table 20.4 and Figure 20.2 analyze factors contributing to changes in manufacturing industry CO_2 emissions over the period 1970–94. While the average annual increase in production by value added between 1974 and 1994 was maintained at a reasonable level of 4.06 per cent, average carbon emissions fell by 0.71 per cent. Table 20.4 and Figure 20.2 indicate that 71 per cent of this reduction in carbon emissions can be attributed to efforts to improve energy efficiency

Table 20.4 Factors contributing to change in CO_2 emissions in the Japanese manufacturing industry, 1970–94

Period	CO_2 emissions $\frac{\Delta CO_2}{CO_2}$	Fuel switching $\frac{\Delta C/E}{C/E}$	Energy efficiency $\frac{\Delta E/Y}{E/Y}$	Change in ind. struct. $\frac{-\Delta V/Y}{V/Y}$	GDP growth $\frac{\Delta V}{V}$	Miscellaneous ε
1970–73	**7.12**	−2.19	−0.20	−1.10	11.00	−0.39
1974–78	**−2.29**	−0.24	−3.98	−0.32	2.31	−0.06
1979–82	**−4.11**	−1.09	−6.31	−2.93	6.34	−0.12
1983–86	**0.11**	1.28	−5.00	−0.13	4.27	−0.31
1987–90	**2.60**	−0.72	−2.66	−1.83	8.04	−0.23
1991–94	**0.52**	−0.15	1.10	−0.13	−0.24	−0.06
1974–94	**−0.71**	−0.19 (4.0%)	−3.40 (71.3%)	−1.03 (21.6%)	4.06	−0.15 (3.1%)

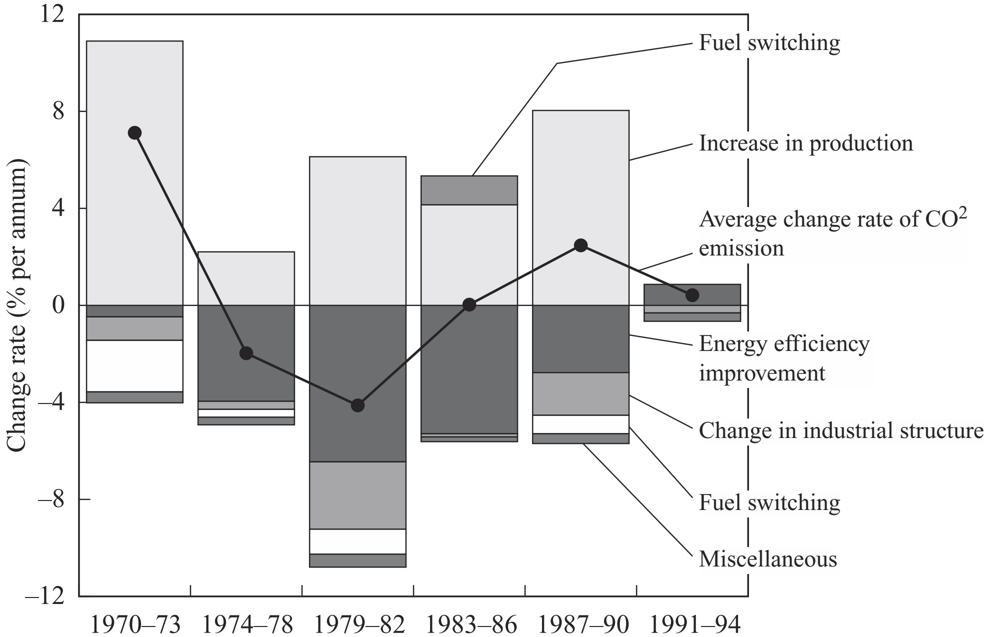


Figure 20.2 Trends in factors and their magnitude contributing to change in CO_2 emissions in the Japanese manufacturing industry, 1970–94

($\Delta(E/Y)/(E/Y)$), while 22 per cent can be attributed to a change in industrial structure. Fuel switching ($\Delta(C/E)/(C/E)$) contributed only 4 per cent. This analysis confirms that Japan's success in continuing economic development after the first energy crisis in 1973 depended largely on the results of coordinated industry–government efforts to reduce energy dependency by increasing efficiency.

If we look carefully at these trends, we note that carbon emissions increased after 1983 (the start of the fall of international oil prices) owing to an increase in coal dependency and a decrease in energy efficiency improvement efforts. Since 1987 (the start of Japan's so-called 'bubble economy') energy efficiency improvement efforts have significantly decreased, also leading to higher carbon emissions. Although carbon emissions decreased again after 1991, this was due solely to a decrease in the GDP as a result of the collapse of the 'bubble economy.' Meanwhile energy efficiency improvement efforts have continued to decline.

As stated earlier, dramatic improvement in energy efficiency in Japan from the late 1970s can be attributed largely to technological innovation. To analyze this situation more thoroughly we need an economic description of technological innovation. Hogan and Jorgenson (1991) stressed the significance of the description of technology change in energy-economic models and made extensive efforts to endogenize technological change using the trans-log production function. However, their efforts to explain technical change in terms of base year prices were unsatisfactory for several reasons, including the lack of a theory of technological change. Nevertheless, they suggested that the common economic growth model assumption of constant technology, or even exogenous technological change (Solow 1957), could be de-emphasized or even eliminated in energy-economic models. A number of authors have investigated such models, using a production function in which energy is incorporated in a standard production function together with capital and labor (for example, Hannon and Joyce 1981; Kümmel 1982b). Such functions have yielded quite good 'explanations' of GNP growth, although most economists do not like them because they seem to contradict the economic theory of income distribution which implies that energy resource owners 'should' be receiving a large fraction of the national income (comparable to returns to labor and capital). This is clearly not the case.

Up to now, no economic growth model in the economics literature has incorporated an explicit theory of technical change. The model described hereafter attempts to fill this gap. As a starting point, note that the change in energy efficiency ($\Delta E/Y$) reflects a dynamic relationship between changes in energy consumption (or demand) (ΔE) and aggregate production (ΔY). Technology (T), defined as a stock of knowledge, obviously has a significant impact on changes in both energy demand and aggregate production: improved technology contributes to increasing production while reducing energy consumption. Hereafter, technology is subdivided into non-energy technology (TnE) and energy technology (TE). While the former aims primarily at increasing either the quantity or quality of goods and services produced, the latter aims at both energy conservation and supply-oriented technologies. In the Japanese case, it focuses primarily on minimizing dependence on imported oil.

In order to undertake a quantitative model analysis, we need a quantifiable measure of both energy technology and non-energy technology. This has been obtained by calculating the 'stock' of knowledge resulting from accumulated expenditures on energy R&D and non-energy R&D, subject to depreciation losses (due to obsolescence). The specific scheme employed is as follows (Watanabe 1992a, 1996a). Let

$$\begin{aligned} T_t &= R_{t-mt} + (1-\rho_t)T_{t-1}, \\ \rho_t &= \rho\,(T_t), \\ m_t &= m(\rho_t), \end{aligned} \tag{20.2}$$

where T_t is the technology knowledge stock in the period t, R_t is the R&D expenditure in the period t, m_t is the time lag of R&D to commercialization in the period t and ρ_t is the rate of obsolescence of technology in the period t.

Next, using equation (20.2), trends in the technology knowledge stock of both energy R&D and non-energy R&D in the Japanese manufacturing industry over the period 1965–94 were measured, as summarized and illustrated in Table 20.5 and Figure 20.3. From the table and figure it can be seen that the priority of R&D shifted from non-energy R&D to energy R&D from the beginning of the 1970s, in the Japanese manufacturing industry. This trend reflects the economic impact of the energy crises in 1973 and 1979, and expenditure on energy R&D rapidly increased, particularly between 1974 and 1982. However, after international oil prices started to fall in 1983, energy R&D expenditure decreased dramatically.

Corresponding to these trends, with a certain amount of time lag, the technology knowledge stock of energy R&D increased dramatically during the period 1974–82. It continued to increase in the period 1983–6, but changed to a sharp decline from 1987 on.

The rapid increase in the technology knowledge stock of energy R&D over a limited

Table 20.5 Trends in change rate of R&D expenditure and technology knowledge stock in the Japanese manufacturing industry, 1970–94 (% per annum)

	R&D expenditure (fixed price)		Technology knowledge stock		
	Total R&D	Energy R&D	Total stock	Stock of energy R&D	Stock of non-energy
1960–69	15.78	9.83			
1970–73	9.91	16.56	16.00	10.58	16.08
1974–78	3.16	20.82	12.51	15.57	12.47
1979–82	9.65	25.44	5.89	24.31	5.55
1983–86	11.30	0.04	6.81	14.31	6.59
1987–90	8.16	0.90	8.16	3.84	8.31
1991–94	−0.70	0.46	7.76	2.25	7.91

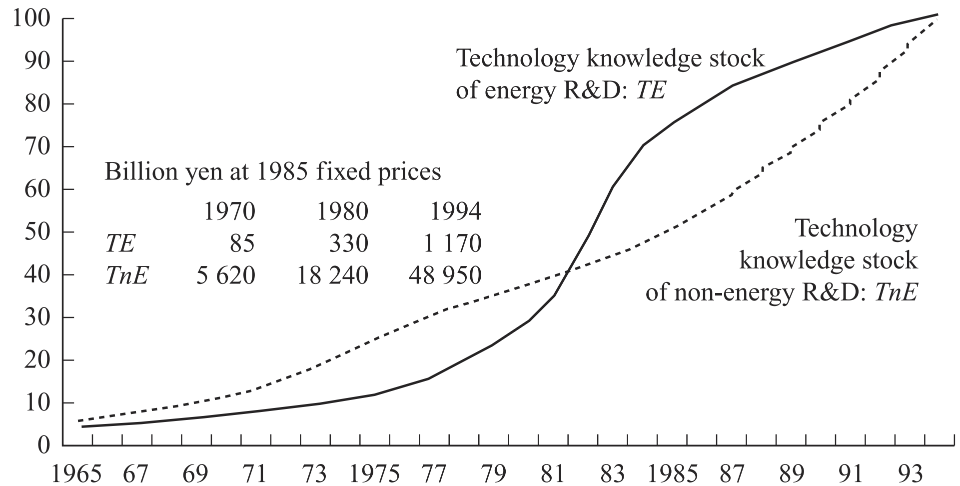


Figure 20.3 Trends in technology knowledge stock of energy R&D and non-energy R&D in the Japanese manufacturing industry, 1965–94. Index: 1965 = 1; 199 = 100

period (1974–86) resulted in a rapid increase in the rate of technology obsolescence, or depreciation. Obsolescence increased from 15.4 per cent in 1974 to 21.2 per cent in 1987. This, in turn, resulted in a rapid decrease in the time lag between R&D and commercialization (which decreased from 3.4 years in 1974 to 1.4 years in 1987).

Provided that technology (T) is embedded in other factors of production (Y) (namely labor, L; capital, K; materials, M; and energy, E) to production (Y), the production function can be written as

$$Y = F(L_{(T)}, K_{(T)}, M_{(T)}, E_{(T)}). \tag{20.3}$$

Then the change rate of energy efficiency ($\Delta(E/Y)/(E/Y)$ where $\Delta(E/Y) = d(E/Y)/dt$) can be calculated as

$$\frac{\Delta(E/Y)}{(E/Y)} = \sum \frac{\partial Y}{\partial X} \times \frac{X}{Y} \times \frac{\Delta(E/X)}{(E/X)} \quad (X = L,K,M). \tag{20.4}$$

E/*X* is a ratio of energy and other services of input. Provided that *E*/*X* is governed by the ratio of prices of respective services of input and energy (*Px*/*Pe*) and technical change (λ_t, where *t* indicates the time trend) (Binswanger 1977), *E*/*X* can be estimated as

$$E/X = E/X\ (Px/Pe,\ \lambda t), \tag{20.5}$$

where *Pe* and *Px* (=*Pl*, *Pk*, *Pm*) are prices of energy, labor, capital and materials, respectively.

Now we decompose λ_t into improvements from an increase in the technology knowledge stock of both energy R&D (*TE*) and non-energy R&D (*TnE*) generated by R&D investment, and other improvements with a linear function of time derived from such effects as scale of economies and learning effects (λ_t). Equation (20.5) can be estimated by the following function for the Japanese manufacturing industry over the period 1974–94

$$\ln E/X = a + b_1 \ln (Px/Pe) + b_{21} \ln TE + b_{22} \ln TnE + \lambda t. \tag{20.6}$$

Under the assumption that the production function is linear and homogeneous, and that prices of respective services of input are decided competitively, by synchronizing equations (20.4) and (20.5) the change rate of energy efficiency can be calculated as

$$\frac{\Delta(E/Y)}{(E/Y)} = \sum \frac{GXC}{GC} \left[b_1 \frac{\Delta(Px/Pe)}{(Px/Pe)} + b_{21} \frac{\Delta TE}{TE} + b_{22} \frac{\Delta TnE}{TnE} + \lambda' \right], \tag{20.7}$$

where *GC* stands for gross cost, and *GXC* stands for gross cost of *X*.

The results of the calculation are summarized and illustrated in Table 20.6 and Figure 20.4. From the table and figure we note that Japan's manufacturing industry's achievement of a 3.4 per cent average annual improvement in energy efficiency over the period 1974–94 can be attributed to the following components: 55.4 per cent to energy technology, 24.9 per cent to non-energy technology, 8.0 per cent to other efforts in response to the sharp increase in energy prices, and 11.7 per cent to non-technology-oriented autonomous energy efficiency improvement derived from such effects as scale of economies and learning effects.

The above analysis supports the previous hypothesis that Japan, in the face of the damaging impacts of the energy crises, made every effort to substitute a constraint-free (or unlimited) production factor, technology, for a constrained (or limited) production factor (energy), as its survival strategy. However, if we look carefully at these trends, we note that the contribution of energy technology, the main contributor to energy efficiency improvement, has decreased since 1983 (the start of the fall of international oil prices). Furthermore, this decrease accelerated from 1987 (the start of Japan's 'bubble economy') and accelerated further from 1991 (the start of the bursting of the 'bubble economy'). This development was the main source of the deterioration in energy efficiency improvement, resulting in an increase in CO_2 discharge, as analyzed in Table 20.4 and Figure 20.2.

The foregoing analysis offer a warning that, despite its success in overcoming energy and environmental constraints in the 1960s, 1970s and the first half of the 1980s, Japan's

Table 20.6 Factors contributing to change in energy efficiency in the Japanese manufacturing industry, 1970–94 (% per annum)

Period	$\frac{\Delta E/Y}{E/Y}$	Labor L	Capital K	Materials M	Miscel. ε	Contribution factors Pe/Px	TE	TnE	λ	ε
1974–78	**−3.98**	0.09	−0.85	−3.80	0.58	−1.53	−2.90	−1.34	−0.17	1.96
1979–82	**−6.31**	−0.58	−1.10	−5.05	0.42	−1.39	−4.96	−0.74	−0.25	1.03
1983–86	**−5.00**	−0.32	−1.13	−4.40	0.85	0.33	−3.46	−1.23	−0.36	−0.28
1987–90	**−2.66**	0.51	−0.95	−3.10	0.88	0.85	−0.99	−1.69	−1.10	0.27
1991–94	**1.10**	0.66	−0.20	−0.75	1.39	0.20	−0.38	−0.71	−0.94	2.93
1974–94	**−3.40**	0.08	−0.85	−3.44	0.81	−0.37	−2.56	−1.15	−0.54	1.22
						(8.0%)	(55.4%)	(24.9%)	(11.7%)	

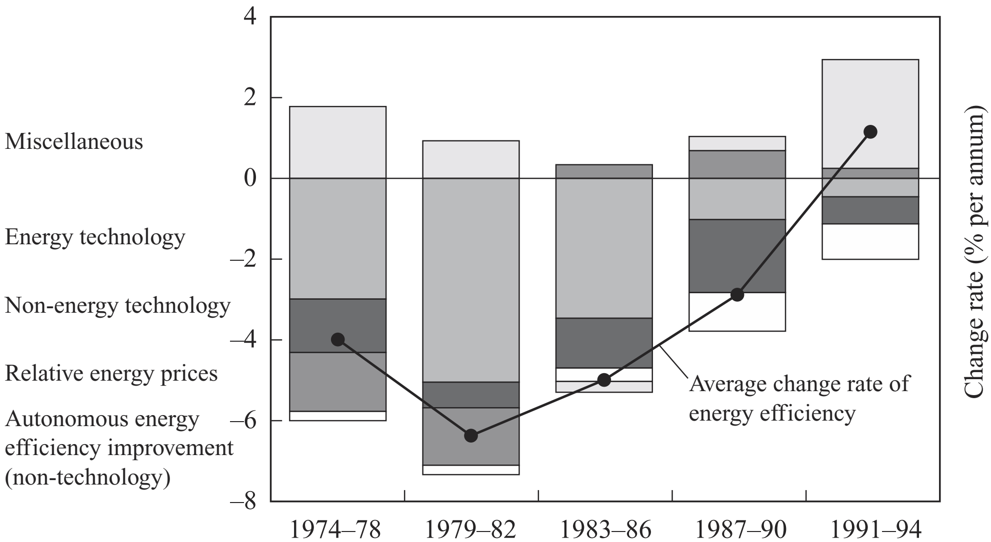


Figure 20.4 Factors contributing to change in energy efficiency in the Japanese manufacturing industry, 1970–94

economy once again faces the prospect of energy and environmental constraints following the fall of international oil prices, the subsequent 'bubble economy' and its eventual collapse (Industrial Technology Council of MITI 1992). The slow (or negative) growth since 1990 is closely correlated to the sharp decline in energy-related R&D.

In order to test this conjecture quantitatively, equation (20.8) analyzes factors governing the Japanese manufacturing industry's energy R&D expenditure over the period 1974–94:

$$\ln ERD = -6.57 + \underset{(6.81)}{6.57 ln\ MERD} + \underset{(3.54)}{0.27 ln(MnERD)} + \underset{(3.32)}{0.74 lnRD} + \underset{(4.10)}{0.64 lnMe} + \underset{(2.26)}{0.25 Pet} \quad (20.8)$$

adj. R^2 0.993 DW 2.07

where *ERD* and *RD* stand for manufacturing industry's energy R&D and total R&D expenditure; *MERD* and *MnERD* stand for MITI's energy R&D and non-energy R&D budget; *Me* is the time lag between energy R&D and commercialization; and *Pet* stands for relative energy prices with respect to capital prices of technology.

Equation (20.8) corroborates earlier findings that MITI's energy R&D budget, together with industry's own total R&D, exerts a strong influence over manufacturing industry's energy R&D expenditure. This also supports the previous analyses showing that MITI's energy R&D benefits industry's energy R&D. In addition to these factors, equation (20.8) indicates that manufacturing industry's energy R&D is sensitive to a time lag between energy R&D and commercialization. Therefore R&D decreases as this time lag decreases. This demonstrates that industry's profitable energy R&D seeds have been depleted owing to a tempered undertaking in a limited period, much like a local rainstorm. Other factors comprised by equation (20.8) include MITI's non-energy R&D budget and relative energy prices with respect to capital technology.

Table 20.7 and Figure 20.5 summarize and illustrate the result of an analysis of factors contributing to the decrease in manufacturing industry's energy R&D expenditure. The table and figure indicate that decreases in MITI's energy R&D budget, industry's total R&D expenditure and the time lag between energy R&D and commercialization are major sources of the stagnation of manufacturing industry's energy R&D from 1983.

Table 20.7 Factors contributing to change in energy R&D expenditure in the Japanese manufacturing industry, 1974–94 (% per annum)

Period	Industry energy R&D $\frac{\Delta ERD}{ERD}$	MITI energy R&D $\frac{\Delta MERD}{MERD}$	MITI non-energy R&D $\frac{\Delta MnERD}{MnERD}$	Industry total R&D $\frac{\Delta RD}{RD}$	Time lag of R&D to comm. $\frac{\Delta Me}{Me}$	Relative energy prices $\frac{\Delta Pet}{Pet}$	Miscellaneous ε
1974–78	31.77	20.33	1.08	8.19	−3.01	6.44	−1.26
1979–82	32.99	20.64	6.22	11.36	−7.34	2.25	−0.14
1983–86	−0.09	2.37	2.15	7.73	−8.70	−2.29	−1.35
1987–90	3.46	1.78	1.03	7.88	−4.25	−2.04	−0.94
1991–94	0.35	−1.08	2.32	−0.50	−3.39	1.33	1.67
1974–94	14.56	9.36	2.49	6.99	−5.23	1.39	−0.44

Note: *ERD* and *RD*: manufacturing industry's energy R&D and total R&D expenditure; *MERD* and *MnERD*: MITI's energy R&D and non-energy R&D budget; *Me*: time lag of energy R&D to commercialization; *Pet*: relative energy prices with respect to capital prices of technology; and ε: miscellaneous.

IMPLICATIONS FOR SUSTAINABLE DEVELOPMENT

Increasing energy and environment constraints, especially the global environmental consequences of energy use, are causing mounting concern around the world. It is widely thought that such constraints may limit future economic growth. In this context, Japan's success in overcoming the energy crises while maintaining economic growth, in the 1970s

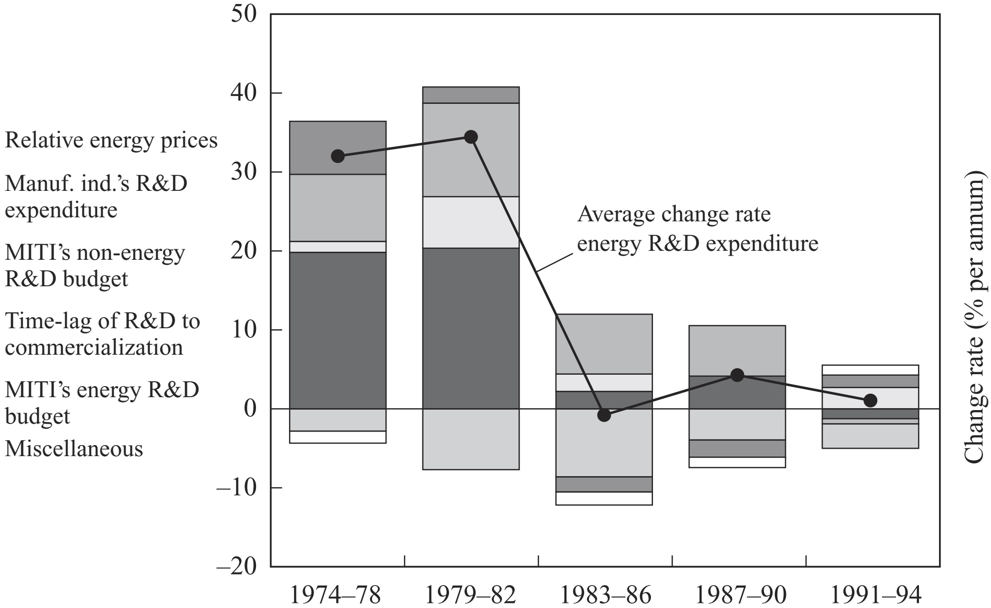


Figure 20.5 Factors contributing to change in energy R&D expenditure in the Japanese manufacturing industry, 1974–94

and 1980s, through a policy of encouraging technological innovation in energy efficiency could provide useful clues for other countries and regions.

In light of this, the systems options for the rational use of energy on the global scale have become crucial. The options can be identified to find the most effective combination of energy efficiency improvement and fuel switching (and also carbon sequestration in the future). The complexity of the global environmental consequences results from the heterogeneity of economic, industrial, geographical, social and cultural conditions of each respective country or region. This implies that we cannot expect any uniform solution at the global level. Nevertheless, we can expect to uncover many opportunities and comparative advantages, in some of which every country/region can share. It is hoped that we can also anticipate some broad-based systems options (for example, decarbonization, dematerialization) and the possibility of realizing a maximum multiplier effect by synchronizing comparative advantages in a systematic way. Given that the global environmental issue is a problem common to all countries of the world, we should seek ways of maximizing the multiplier effect. A comprehensive systems approach is therefore critical.

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PART IV

Industrial Ecology at the National/Regional Level

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21. Global biogeochemical cycles

Vaclav Smil

The importance of global biogeochemical cycles is easily stated: all economic systems are just subsystems of the biosphere, dependent on its resources and services. The biosphere cannot function without incessant cycling of scarce elements needed for prokaryotic and eukaryotic metabolism.

The water cycle is the biosphere's most rapid and the most massive circulation. It is driven overwhelmingly by evaporation and condensation. Compared to the ocean, the living organisms have only a negligible role in storing water, and they are of secondary importance in affecting its flows. (Evapotranspiration supplies only about 10 per cent of all water entering the atmosphere.) Human activities have drastically changed some local and even regional water balances and pronounced anthropogenic global warming would accelerate the global water cycle. But, with the exception of globally negligible withdrawals from ancient aquifers, and water vapor from combustion, we do not add to the compound's circulating mass.

In contrast, human activities – above all the combustion of fossil fuels – have been introducing large amounts of carbon (C), nitrogen (N) and sulfur (S) into the biosphere. These elements are doubly mobile, being transported in water as ionic solutions or in suspended matter, and through the atmosphere as trace gases. Theirs are the three true biospheric cycles as they are dominated by microbial and plant metabolism. They involve numerous nested subcycles, and operate on time scales ranging from minutes to millions of years, as the elements may move rapidly among reservoirs or be sequestered (assimilated, mineralized, immobilized) for extended periods of time.

High mobility of C, N and S makes the three elements readily available in spite of their relative biospheric scarcity. But it also means that human interference in these cycles has become evident on the global level (for example, rising atmospheric concentrations of CO_2, CH_4 or N_2O) and/or that it has major impacts on large regional or continental scales (atmospheric deposition of sulfates and nitrates). Environmental problems arising from these changes include potentially rapid global warming, widespread acidification of soils and waters, and growing eutrophication of aquatic and terrestrial ecosystems. These topics have been receiving a great deal of research attention in recent years (Turner *et al.* 1990; Schlesinger 1991; Butcher *et al.* 1992; Wollast *et al.* 1993; Mackenzie and Mackenzie 1995; Smil 1997; Agren and Bosatta 1998).

Finally, those mineral elements that form no, or no stable, atmospheric gases are moved through the biosphere solely by the sedimentary–tectonic cycle. Weathering liberates these elements from parental materials and they travel in ionic solutions or as suspended matter to be eventually deposited in the ocean. They return to the biosphere only when the reprocessed sediments re-emerge from ancient seabeds or from the mantle in ocean ridges or hot spots. We thus see mineral cycles only as one-way oceanward flows, with human activities (mineral extraction, fuel combustion) enhancing some of these fluxes, particularly by mobilization of heavy metals.

This chapter reviews the basics of C, N and S cycles. Because of the element's indispensable role in food production and the intensity of its anthropogenic mobilization, it also looks briefly at phosphorus (P) flows in the biosphere. In every case it stresses the extent of recent human interference – the essence of industrial ecology.

CARBON

At about 280 ppm the preindustrial atmospheric CO_2 amounted to almost exactly 600 billion tonnes (Gt) C. The ocean contained roughly 37000Gt of dissolved inorganic C, about 95 per cent of it as bicarbonate ion (HCO^{-3}). The fate of marine C is controlled by interactions of physical, chemical and biotic processes. Amounts of dissolved inorganic C depend on the release of CO_2 to the atmosphere and on its equilibrating uptake; on precipitation and dissolution of marine carbonates; and on photosynthetic assimilation and organic decay. The equilibrium absorptive capacity of the ocean is a function of temperature and acidity (pH). Moreover, it becomes available only after the whole water column equilibrates with the added CO_2, a process limited by the ocean's layered structure. Intermediate and deep waters, beyond the reach of solar radiation with temperatures at 2–4 °C, contain nearly 98 per cent of the ocean's C.

Photosynthesis in the ocean assimilates annually about 50Gt C, turning circumpolar oceans into major C sinks during the summer months. Respiration by zooplankton, and by other oceanic herbivores, returns at least 90 per cent of the assimilated carbon to the near-surface waters from which the gas can either escape to the atmosphere or be re-used by phytoplankton. Carbon in the remaining dead biomass settles to the deeper ocean. In reversing the photosynthetic process, this settling, oxidation and decay increases the ocean's total carbon content while slightly lowering its alkalinity (increasing acidity).

Pre-agricultural terrestrial photosynthesis took up annually about twice as much C as did marine phytoplankton. Both C storage and net C exchange of ecosystems are dependent on photosynthetically active radiation. Plant C uptake is marked by pronounced daily and seasonal cycles, the latter fluctuations producing unmistakable undulations of the biospheric 'breath'. Rapid recycling of Plant C was almost equally split between autotrophic and heterotrophic respiration. Decomposition of organic litter (dominated by bacteria, fungi and soil invertebrates) and herbivory remain the two most important forms of heterotrophic respiration.

Soils contain more than twice as much C as the atmosphere, and their C storage density goes up with higher precipitation and lower temperature. Accumulation of a tiny fraction of assimilated C in sediments is the principal terrestrial bridge between the element's rapid cycle – whereby decomposition of biomass returns the assimilated C into the atmosphere in just a matter of days or months so that it can be re-used by photosynthesis – and slow cycles which sequester the element in breakdown-resistant humus or in buried sediments. Long-term exposure of the sequestered organic C compounds (for up to 10^8 years) to higher temperatures and pressures results in the formation of fossil fuels whose global resources are most likely in excess of 6 trillion tonnes C. Only a small part of these fossil fuels will be eventually recovered by our industrial civilization.

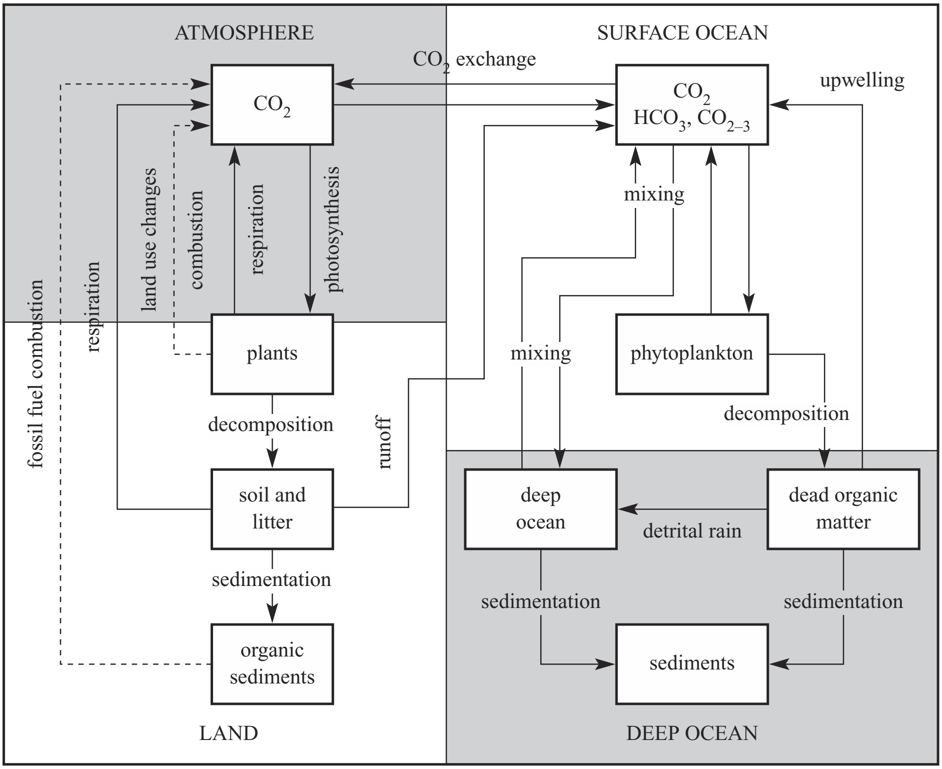


Figure 21.1 Global carbon cycle

Extraction of these fossil fuels has been emitting increasing amounts of C into the atmosphere. The annual global rate rose from less than 0.5Gt C in 1900 to 1.5Gt C in 1950 and by the year 2000 it surpassed 6Gt C. Additional net release of 1–2Gt C comes from ecosystems converted to other uses (mainly from tropical deforestation). The atmospheric CO_2 trend is reliably known for nearly half a million years thanks to the analyses of air bubbles from Antarctic and Greenland ice cores. During this period CO_2 levels have stayed between 180 and 300 ppm. During the 5000 years preceding 1850 they had fluctuated just between 250 and 290 ppm.

When the first systematic measurements began in 1958, CO_2 concentration averaged 320 ppm. In the year 2000 the mean at Mauna Loa surpassed 370 ppm. A nearly 40 per cent increase in 150 years is of concern because CO_2 is a major greenhouse gas whose main absorption band coincides with the Earth's peak thermal emission. Greenhouse effects maintain the Earth's average surface temperature at around 15 °C, or about 33 °C warmer than would be the case otherwise. Thermal energy reradiated to Earth by the atmosphere, 325W/m^2 (Watts per square meter), is the most important source of heat for oceans and continents, and current atmospheric levels of CO_2 contribute about 75W/m^2.

Anthropogenic CO_2 has already increased the heat flux by 1.5W/m^2, and higher levels of other infrared (IR) absorbing gases have brought the total forcing to about 2.5 W/m^2. Rising emissions would eventually lead to the doubling of preindustrial greenhouse gas levels and to average tropospheric temperatures 1–5 °C higher than today's mean. This warming would be more pronounced on the land and during the night, with winter increases about two to three times the global mean in higher latitudes than in the tropics, and greater in the Arctic than in the Antarctic.

Major worrisome changes arising from a relatively rapid global warming would include intensification of the global water cycle accompanied by unequally distributed shifts in precipitation and aridity; higher circumpolar run-offs, later snowfalls and earlier snowmelts; more common, and more intensive, extreme weather events such as cyclones and heat waves; thermal expansion of sea water and gradual melting of mountain glaciers leading to appreciable (up to 1m) sea level rise; changes of photosynthetic productivity and shifts of ecosystemic boundaries; and poleward extension of tropical diseases (Watson *et al.* 1996). International efforts to reduce the rate of CO_2 emissions have been, so far, unsuccessful: cuts proposed by the Kyoto Treaty (amounting to emissions about 7 per cent below the 1990 rate) were rejected by the USA, the world's largest greenhouse gas producer, and they do not even apply to China, now the second largest contributor of CO_2.

NITROGEN

The atmosphere is nitrogen's largest biospheric reservoir, with stable N_2 molecules forming 78 per cent of its volume. Trace amounts of NO and NO_2 (designated jointly as NOx), of nitrous oxide (N_2O), nitrates (NO^{-3}) and ammonia (NH_3) are also present. The N content of soils varies by more than an order of magnitude, but most of it is embodied in humus. Water stores very little nitrogen: ammonia is not very soluble, and nitrate concentrations in natural (uncontaminated) streams are very low. Most of the plant tissues are N-poor polymers, but N is present in every living cell in nucleic acids, which store and process all genetic information, in amino acids, which make up all proteins, and in enzymes. It is also a component of chlorophyl, whose excitation energizes photosynthesis.

The natural N cycle is driven overwhelmingly by bacteria. Fixation, nitrification and denitrification are the basic pathways of the cycle. Fixation, the conversion of unreactive N_2 to reactive compounds, can be both abiotic and biogenic. Lightning severs the strong N_2 bond, and the element then forms NO and NO_2, which are eventually converted to nitrates. Biofixation, moving N_2 to NH_3, is performed only by bacteria thanks to nitrogenase, a specialized enzyme no other organisms carry. *Rhizobium* is by far the most important symbiotic fixer, forming nodules on leguminous plant roots. There are also endophytic bacteria (living inside plant stems and leaves) and free living N-fixers, above all cyanobacteria.

Nitrifying bacteria present in soils and waters transform NH_3 to NO^{-3}, a more soluble compound plants prefer to assimilate. Assimilated nitrogen is embedded mostly in amino acids which are the building blocks of all proteins. Heterotrophs (animals and people) must ingest preformed amino acids in feed and food in order to synthesize proteins tissues. After plants and heterotrophs die, enzymatic decomposition (ammonification) moves N from dead tissues to NH_3, which is again oxidized by nitrifiers. Denitrification returns the element from NO^{-3} (via NO^{-2}) to atmospheric N_2. However, incomplete reduction results in some emissions of N_2O, a greenhouse gas about 200 times more potent than CO_2.

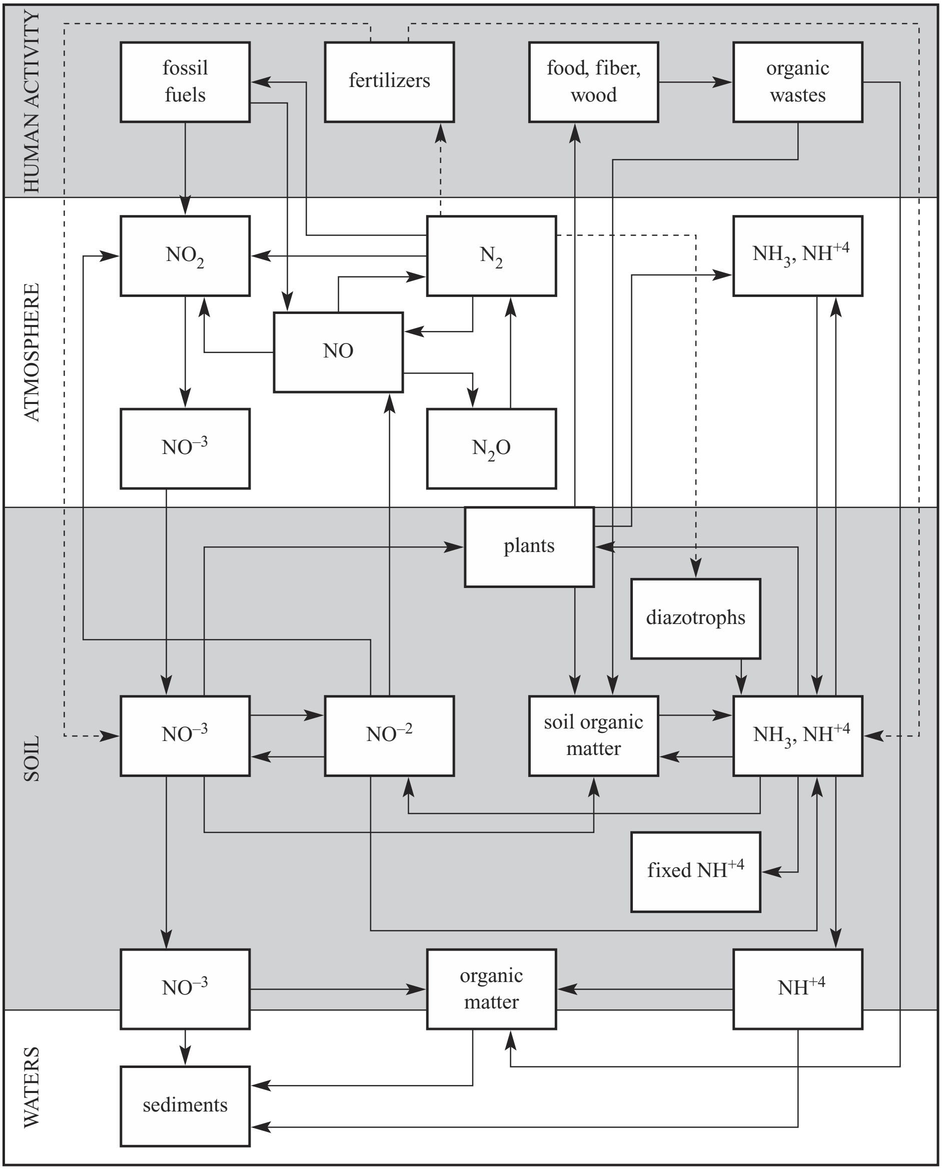


Figure 21.2 Global nitrogen cycle

There are many leaks, detours and backtracking along this main cyclical route. Volatilization from soils, plants and animal and human wastes returns N (as NH_3) to the atmosphere, to be redeposited, after a short residence, in dry form or in precipitation. Both nitrification and denitrification release NOx and N_2O. Nitrogen in NOx returns to the ground in atmospheric deposition, mostly after oxidation to NO_3. In contrast, N_2O is basically inert in the troposphere but it is a potent greenhouse gas, and it contributes to the destruction of stratospheric ozone. Highly soluble nitrates leak readily into ground and surface waters, and both organic and inorganic nitrogen in soils can be moved to waters by soil erosion.

Pre-agricultural terrestrial fixation of N, dominated by biofixation in tropical forests, amounted to at least 150–190 million tonnes (Mt) N/year (Cleveland *et al.* 1999). Planting of leguminous crops, practiced by every traditional agriculture, was the first major human intervention in the cycle. It now fixes annually 30–40 Mt N. Guano and Chilean $NaNO_3$ were the first commercial N fertilizers: their exports to Europe began before 1850. By 1910, by-product ammonia from coking, calcium cyanamide and calcium nitrate from an electric arc process also provided relatively small amounts of fixed N.

Only the synthesis of ammonia from its elements – demonstrated for the first time by Fritz Haber in 1909, and commercialized soon afterwards by the BASF under the leadership of Carl Bosch – opened the way for a large-scale, inexpensive supply of fixed N (Smil 2001). Haber–Bosch fixation expanded rapidly only after 1950 and it became particularly energy-efficient with the introduction of single-train plants equipped with centrifugal processors that were commercialized during the 1960s. The current rate of global NH_3 synthesis surpasses 100Mt N/year. About four-fifths of it is used as fertilizers (mostly as a feedstock for producing urea and various nitrates, sulfates and phosphates). The rest goes into industrial process, ranging from the production of explosives and animal feed to feedstocks for syntheses of dyes, plastics and fibers (for example, nylon; Febre-Domene and Ayres 2001).

Typically no more than half of the N applied to crops is assimilated by plants. The rest is lost owing to leaching, erosion, volatilization and denitrification (Smil 1999). Uptakes are lowest (often less than 30 per cent) in rice fields, highest in well-farmed, temperate crops of North America and Northwestern Europe. Because the primary productivity of many aquatic ecosystems is N-limited, eutrophication of streams, ponds, lakes and estuaries by run-off containing leached fertilizer N promotes growth of algae and phytoplankton. Decomposition of this phytomass deoxygenates water and seriously harms aquatic species, particularly the benthic fauna. Algal blooms may also cause problems with water filtration or produce harmful toxins (for example, 'red tides').

Nitrogen in eutrophied waters comes also from animal manures, human wastes, industrial processes and from atmospheric deposition. There is a clear correlation between a watershed's average rate of nitrogen fertilization and the riverine transport of the nutrient. The worst affected offshore zone in North America is a large region of the Gulf of Mexico, where the nitrogen load brought by the Mississippi and Atchafalya rivers has doubled since 1965, and where eutrophication creates every spring a large hypoxic zone that kills many bottom-dwelling species and drives away fish. Other affected shallow waters include the lagoon of the Great Barrier Reef, and portions of the Baltic, Black, Adriatic and North Seas.

Combustion of fossil fuels is now the source of almost 25Mt N/year as NOx. In large urban areas these gases are essential ingredients for the formation of photochemical smog. Their eventual oxidation to nitrates is a major component (together with sulfates) of acid deposition (see more under 'Sulfur'). Atmospheric nitrates, together with volatilized ammonia, also cause eutrophication of normally N-limited forests and grasslands. In parts of eastern North America, Northwestern Europe and East Asia their deposition (up to 60kg N/hectare per year) has become significant even by agricultural standards

(Vitousek *et al.* 1997). Positive response of affected ecosystems is self-limiting as N saturation leads to enhanced N losses.

SULFUR

Sulfur's critical role in life is mainly to keep proteins three-dimensional. Only two of 20 amino acids providing building blocks for proteins (methionine and cysteine) have S embedded in their molecules. When amino acids form long polypeptide chains, disulfide bridges between two cysteines link them together and maintain their complex folded structure necessary for engaging proteins in countless biochemical reactions.

Sea spray is by far the largest natural source of S entering the atmosphere (140–180Mt S/year). About nine-tenths of this mass is promptly redeposited in the ocean. Volcanoes are a large but highly variable source (mainly as SO_2) with long-term average annual rates of around 20Mt S. Airborne dust, mainly desert gypsum, adds 8–20Mt S/year. Both sulfate-reducing and S-oxidizing bacteria are common in aquatic ecosystems, particularly in muds and hot springs. Biogenic S gases include hydrogen sulfide (H_2S), dimethyl sulfide (DMS), methyl mercaptan and propyl sulfide. H_2S dominates emissions from wetlands, lakes and anoxic soils. Intensity of biogenic emissions increases with higher temperature and their global annual flux amounts to 15–40Mt S/year. Marine biogenic emissions are dominated by DMS from the decomposition of algal methionine.

A possible feedback loop has been postulated between DMS generation and received solar radiation: more DMS would increase cloud albedo, and the reduced radiation would lower planktonic photosynthesis, thus producing fewer condensation nuclei and letting in more insolation. This hypothetical homeostatic control of the Earth's climate by the biosphere was used for some time as an argument for the existence of Lovelock's Gaia. But the magnitude (and indeed the very direction) of the feedback remains questionable (Watson and Liss 1998).

Reduced gaseous S compounds are rapidly oxidized to sulfates and these are deposited back into the ocean or on land in a matter of days. Residence time of SO_2 in humid air may be just a few minutes, and the global mean is approximately one day. Atmospheric H_2S has an equally short residence time, and marine DMS survives about 10 hours before it reacts with OH^-. In turn, sulfates in the lowermost troposphere last usually no longer than 3–4 days. The usual limit of long-distance atmospheric transport (except for S ejected by volcanic emissions all the way to the lower stratosphere) is a few hundred kilometers for SO_2 and H_2S, and between 1000 and 2000 km for sulfates. Consequently, unlike CO_2 or N_2O, atmospheric S does not exhibit a true global cycle.

Wet deposition removes globally some four-fifths of all atmospheric S. The rest is about equally split between dry deposition and SO_2 absorption by soils and plants. Anthropogenic S emissions come overwhelmingly (about 93 per cent) from the combustion of fossil fuels. The remainder originate largely from smelting of metallic sulfides (Cu, Zn, Pb). Global emissions rose from about 5Mt S/year in 1900 to about 80Mt S/year in 2000 (Lefohn *et al.* 1999). Post-1980 decline of S emission in OECD countries (mainly due to conversion from coal to natural gas and desulfurization of flue gases) has been compensated by rising SO_2 generation in Asia, above all in China, now the world's largest consumer of coal.

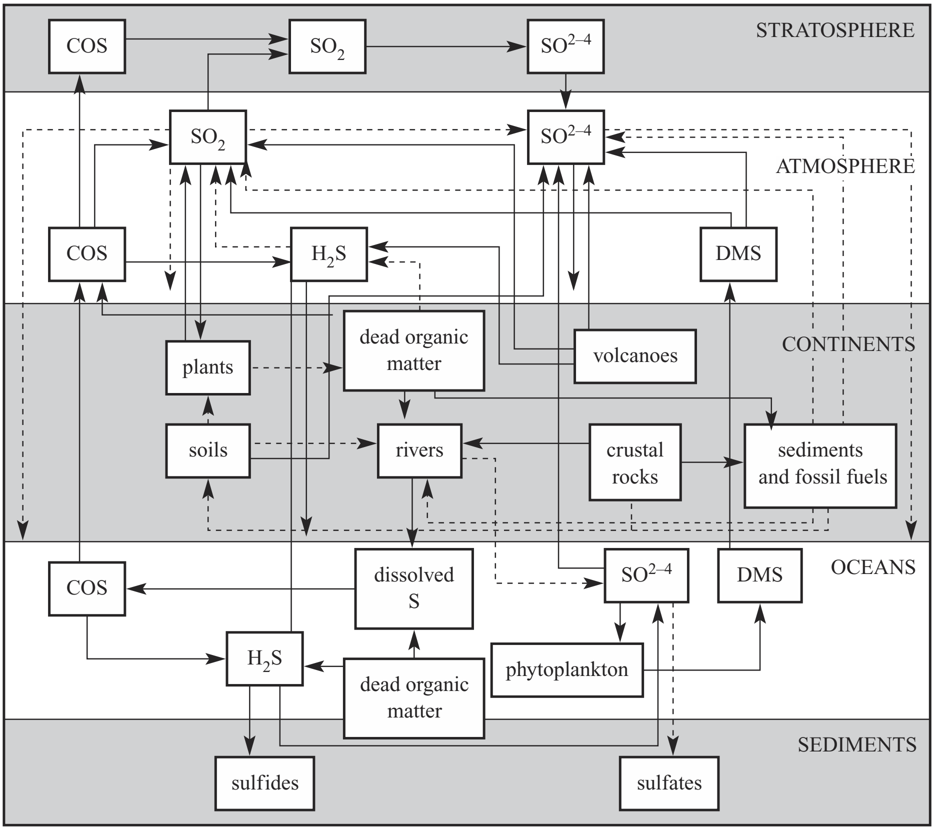


Figure 21.3 Global sulfur cycle

Sulfates in the air partially counteract global warming by cooling the troposphere: the combined global average of natural and anthropogenic emissions is now about $-0.6 W/m^2$. The negative forcing is highest in Eastern North America, Europe and East Asia, the three regions with the highest sulfate levels. Deposited sulfates acidify waters and poorly buffered soils. Extensive research on acid deposition, also including the impact of nitrates, has identified common effects of acidity in aquatic ecosystems devoid of any buffering capacity. These include leaching of alkaline elements and mobilization of toxic Al from soils, and often profound changes in biodiversity of lakes, including the disappearance of the most sensitive fishes, amphibians and insects (Irving 1991). In contrast, the exact role of acid deposition in reduced productivity and dieback of some forests remains uncertain (Godbold and Hutterman 1994).

PHOSPHORUS

Phosphorus (P), rare in the biosphere, is indispensable for life owing to its presence in nucleic acids (DNA and RNA) and in adenosine triphosphate (ATP), the energy carrier for all living organisms. The element is, together with N and K, one of the three macronutrients needed by all crops. Crustal apatites (calcium phosphates) are the element's largest reservoir of the element. Soluble phosphates released by weathering are rapidly transformed to insoluble compounds in soils. As a result, plants absorb P from very dilute solutions and concentrate it up to 1000–fold in order to meet their needs. Thus P released by decomposition of biomass is rapidly recycled.

Rapid P recycling is also the norm in aquatic ecosystems. Even so, the primary productivity in fresh waters, estuaries and particularly in the open ocean is often P-limited. Particulate P that sinks into marine sediments (mainly as calcium phosphates from bones and teeth) becomes available to terrestrial biota only after the tectonic uplift re-exposes the minerals to denudation. The element's global cycle thus closes only after tens to hundreds of millions of years.

In contrast to the studies of C, N and S cycles, there is only a limited amount of global data on P reservoirs and flows (Jahnke 1992; Tiessen 1995; Smil 2000). Terrestrial phytomass stores about 500Mt of the element and plant growth assimilates up to 100Mt P/year. Soils store about 40Gt P, more than four-fifths of it in inorganic compounds. Marine phytomass contains only some 75Mt P but, because of its rapid turnover, it absorbs annually about 1Gt P from surface water. Surface concentrations of P are high only in coastal areas receiving P-rich run-off. These areas contain only about 0.2 per cent of all marine P but they support a disproportionately large share of marine productivity.

Human intensification of biospheric P flows is due to four major processes. Accelerated erosion and run-off due to land use changes now liberate annually more than 20Mt P in excess of the natural loss. Recycling of crop residues returns 1–2Mt P to arable soils, and animal manures return up to 8Mt P every year. The global population of 6 billion people discharges every year about 3Mt P in its wastes; in low-income countries a large share of this is deposited on land, but urbanization puts a growing share of human waste into sewers and then into streams or water bodies. Since the 1940s, P-containing detergents have added another major source of waterborne P.

Inorganic fertilizers represent by far the most important anthropogenic P flux. Their production began during the 1840s with the treatment of P-containing rocks with sulfuric acid. Discovery of large phosphate deposits in Florida (1870s), Morocco (1910s) and Russia (1930s) laid the foundation for a rapid post-1950 expansion of the fertilizer industry. Consumption of P fertilizers peaked at more than 16Mt P in 1988; after a 25 per cent decline by 1993 (due to stagnating grain output and the fall of the USSR) the global use is rising once again. The top three producers (USA, China and Morocco) now account for about two-thirds of the global output. Global food harvest now assimilates about 12Mt P in crops and in their residues, while no more than 4Mt P are supplied by weathering of P-bearing rocks and by atmospheric deposition. Fertilizer P is thus indispensable for producing today's harvests.

In aggregate, human activities are now mobilizing annually more than four times as much P as did the natural processes during the pre-agricultural era. Even relatively low P concentrations present in the run-off from fertilized fields or from sewage can cause eutrophication, or potentiate problems arising primarily from N enrichment. Best field management practices, aimed at reducing P applications, or limiting post-application losses, can be very effective. Coagulating agents (salts of Ca, Mg) remove 70–95 per cent of P in sewage. But microbial processes are cheaper; activated sludge (up to 7 per cent P)

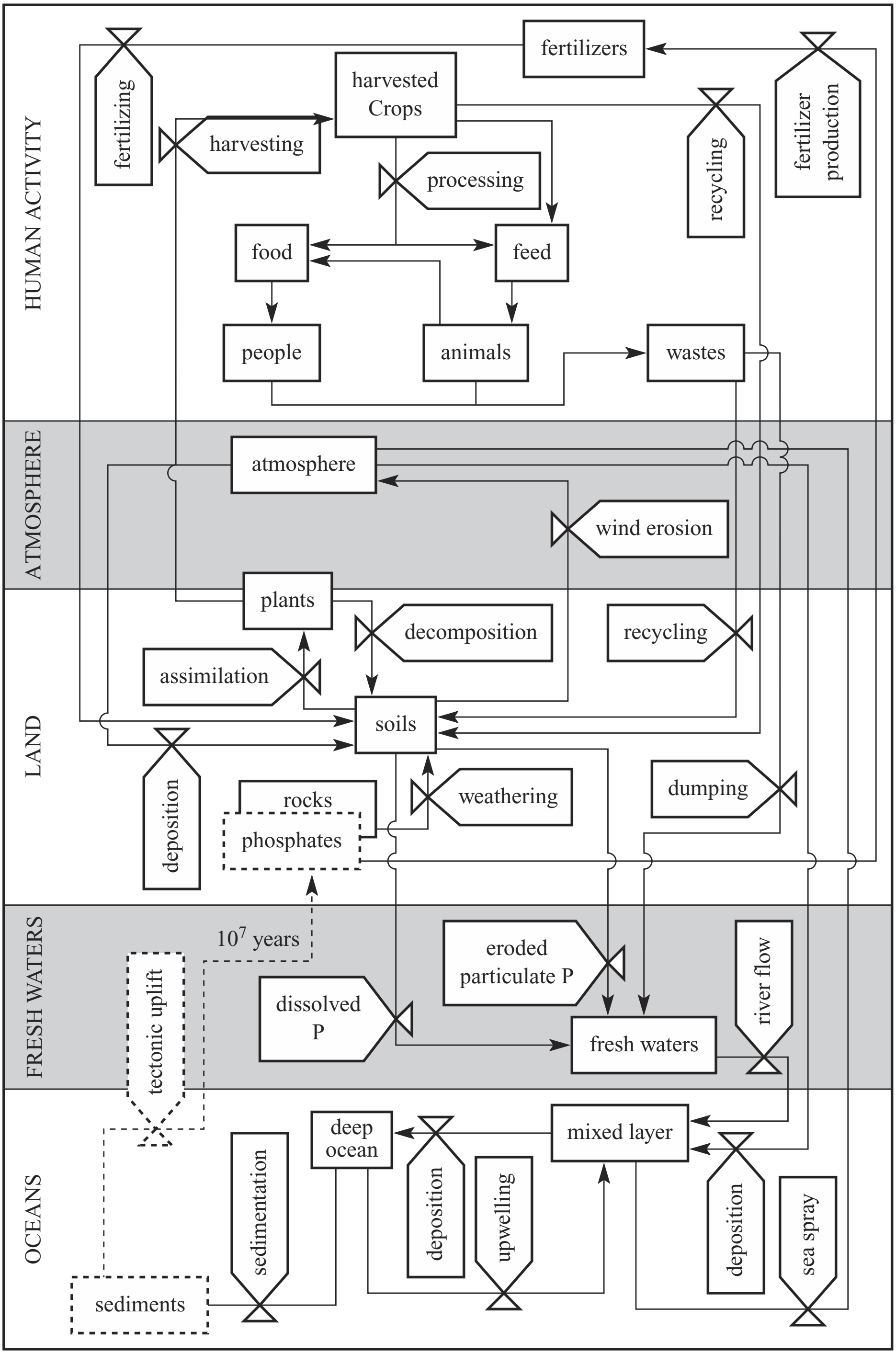


Figure 21.4 Global phosphorus cycle

can be either recycled or dried and incinerated and the nutrient recovered from ash. Yet another environmental problem associated with applications of P fertilizers and recycling of manures and sewage sludges is the presence of cadmium in most phosphate deposits.

HUMAN INTERFERENCE IN BIOGEOCHEMICAL CYCLES

Combustion of fossil fuels and land use changes produce annually about 8Gt C, a small flux compared to natural flows of the C cycle, but the resulting increases of the atmospheric carbon dioxide intensify the Earth's natural greenhouse effect, a process that may eventually result in an unacceptably high rate of global warming. Combined flux of reactive anthropogenic N (from inorganic fertilizers, legume crops and combustion of fossil fuels) now rivals natural terrestrial fixation of the nutrient; consequences of this change range from the stratosphere to coastal waters, with long-term effects of both terrestrial and aquatic eutrophication being most worrisome. Anthropogenic sulfur emissions already surpass the combined flux of biogenic and volcanic S gases, and extensive acidification of sensitive ecosystems is the most important impact of this interference. And human activities have roughly quadrupled the natural mobilization of P, adding to eutrophication problems arising from N enrichment. Fortunately, available technical fixes and socioeconomic adjustments can go a long way towards moderating all of these impacts: what is missing is the commitment to such effective adaptations.

22. Material flow accounts: the USA and the world

Donald G. Rogich and Grecia R. Matos

Movements and transformations, material flows, in the environment are continuous. These can be driven by solar energy and geologic processes, or by living organisms which are part of the natural environment. They can also be the result of human activity. All movements and transformations cause change, and these changes may or may not be compatible with sustaining the environmental conditions that exist. Where changes in one part of an ecosystem are useful to, or reversed by, another component of the same system, the system will remain in balance because the cycle of change is closed. With the exception of energy from the sun, natural systems have closed cycles, things that die and decay, the outputs from one part of the system, produce the nutrients for other living things, which in turn provide the basis for new growth. In contrast, the majority of the outputs from industrial activities have no utility to any other part of the environment, they are wastes, and the cycle of change is open: 'the industrial system is an open one in which nutrients are transformed into "wastes", but not significantly recycled. The industrial system, as it exists today, is therefore *ipso facto* unsustainable' (Ayres and Ayres 1998).

When humans lived as hunter-gatherers they were part of the closed natural system. With the rise of agriculture and the creation of concentrated settlements (cities), they began to live in more open systems, separate at first, but increasingly interconnected. The material flows (movements and transformations of physical material) associated with our emerging open systems grew slowly until the advent of the industrial revolution about 300 years ago, at which time they began to increase exponentially in the countries which were industrializing. Currently, material flows in the USA exceed 20 billion metric tons per year, about 80 tons of material for every person in the country. However, the USA is not alone; studies (Adriaanse *et al.* 1997; WRI 1997) have shown that the material flows in Germany, Japan, the Netherlands and Austria are of equal per capita magnitude. We are busy creatures, and we are changing our environment.

Figure 22.1 presents a conceptual model of material flow in an industrial economy. In this representation inputs are obtained from the domestic environment, and outputs are returned to it. These actions modify the domestic environment. Imports from, and exports to, other countries affect the environment in the country where they are created and disposed. The inputs from the environment are renewable or non-renewable (created in geologic time) resources and pre-existing landforms. Landforms, which are modified to increase their economic utility, change the environment in a 'permanent' manner. Resource inputs extracted from the environment move through a material cycle to a stage where they are ready for use. At each point in this cycle there are process outputs (some of these flows may be recaptured, on the basis of economic and technical considerations) and losses which are a consequence of inefficiencies in the process. After going through all the necessary processing stages the material enters the use stage, where the residence time can vary considerably depending on the material and its use. Some flows result in 'permanent' additions to the stock of built infrastructure; others are discarded after only days or years of useful life. The use of materials such as fertilizers and pesticides results in an immediate, dissipative, release to the environment. As shown in Figure 22.1, some process wastes and post-use discards may be recaptured, and re-used as inputs. The recapture of material flows emitted in process is not always feasible, and those used in a dissipative manner are not recoverable. While in economic terms the value added to the material is consumed by use, the physical material, often in a changed form, continues to exist after it exits from the economy.

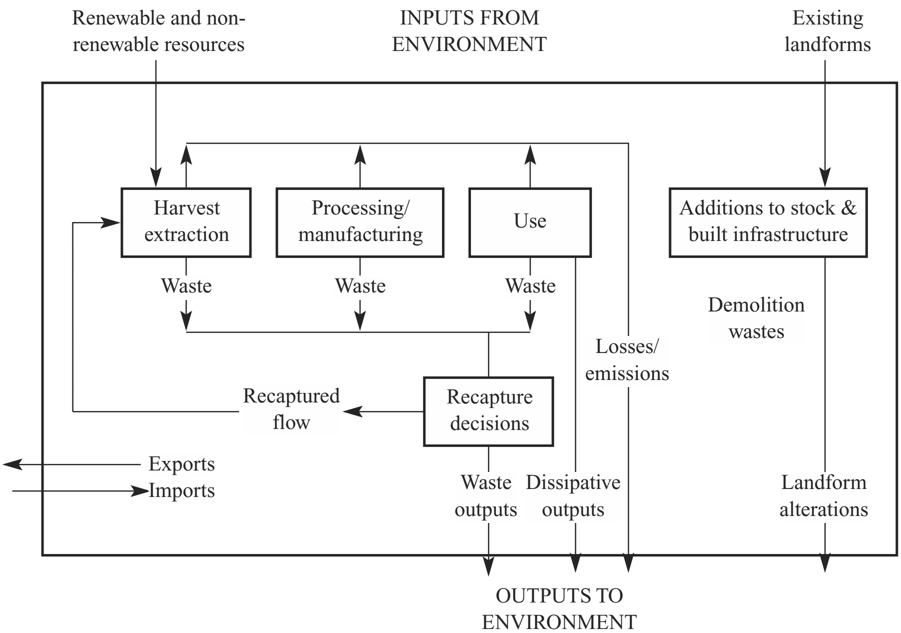


Figure 22.1 The materials cycle

DATA AND METHODOLOGICAL ISSUES

Obtaining a complete picture of the industrial metabolism of an economy requires a thorough understanding of all the material flows. In addition to knowing what the quantities of all the specific flows are, it is important to know something about their residence time in the economy, and the form and mode of output to the environment.

In the USA, data on the quantity of material flow at the first point in the material where an economic transaction takes place is quite good. This is the point where a specific flow may be considered to be a commodity. Examples of the first commodity stage are refined copper, aluminum and lead, clean sand and gravel, forest products, such as lumber, plywood and veneers, fuels such as clean coal, and crude oil delivered to refineries, and agricultural seeds and fibers. Many commodities are also the source of derivative commodities that appear at some point in the material-processing cycle. Examples of these are the various kinds of paper derived from wood and other inputs, and asphalt from crude oil. Data on these commodities are generally available but may become increasingly difficult to obtain as commodities continue to spawn additional products. Table 22.1 lists the specific principal and derivative commodities considered for this chapter.

With the exception of data on recycled quantities, estimates of the flows prior and subsequent to the use phase are scarce and tenuous. Because upstream flows are not normally priced in the economic system, their quantities are hidden from the view of national statistics. Hidden flows include the overburden and concentration waste associated with the mining and initial processing of minerals and metals, harvest waste from the extraction of renewable resources, and the erosion that is a consequence of agriculture and forest activities and other human endeavors. An additional material flow in this category is the transformation of the landscape to accommodate roads and other manifestations of the built infrastructure. Estimates of hidden flows are mostly derived using scattered point estimates and technical judgments.

Comprehensive data on losses, emissions and wastes associated with the processing of commodities and the manufacture of products are also scarce. The EPA Toxic Release Inventory provides data on selected materials that are released, but these make up only a small portion of the total. Estimates of losses and emissions upstream from use are for the most part based on the engineering judgment of technical experts, and material balances, if these can be created.

Specific data on the ultimate fate of material flows that enter use phase are also almost totally lacking. A considerable portion of the flows, those used for construction of the built infrastructure, can remain and provide utility in the economy for a long time, over hundreds of years. All the other flows, with the exception of flows that are recycled, become outputs to the environment after some use period. The length of time before a flow exits from the economy, the mode of the output to the environment (solid, liquid, gaseous) and whether the output is controlled are generally unknown, and must be inferred from information on material use. As an example, salt mined as rock salt in the USA, can be used as mined for de-icing roads, in a processed form as a food additive, or converted into chlorine and caustic soda to provide the feedstock for a complex array of chemicals and physical goods. While the starting point of each of these uses may be the same, the processes used to prepare the product for use, the use of the product, the retention time in the economy, and the character and mode of all the outputs are different. While data from the USA Environmental Protection Agency (EPA) on municipal and construction-and-demolition solid waste are useful, they cannot always be related to specific inputs, and they represent only a small portion of the outputs that occur post-use.

Analyses of flows of water and air, which are used and transformed in the economy, represent a major challenge. In some cases these flows become part of the commodity, polymers being an example. In the case of CO_2 created during fuel combustion, they are incorporated when the commodity is used. However, in other cases they are used for one purpose, for instance cooling, and used again, possibly for irrigation. Most large-scale MFA studies ignore flows of air and water, except where they are part of specific activities of interest, for example, the combustion of fuels (Matthews *et al.* 2000). Exceptions to this are the detailed, one point in time, studies of industrial sectors provided in Ayres and Ayres (1998). In many cases, MFA studies of specific industrial processes may also include flows of water and air.

The time phasing of input and output flows in MFA studies has not been attempted, for the most part. Generally flows are counted as if they enter and leave the economy simultaneously. The residence time in the economy of input flows was characterized using a three-level categorization scheme in Matthews (*et al.* 2000), but outputs were still all accounted for in the same year as the inputs. Simultaneous input and output obviously does not occur, but a case can be made that, where the industrial metabolism of a country is not undergoing radical shifts, the distortion from reality is not unacceptable.

A final methodological issue relates to the quality of output flows, and their potential impacts on the environment. As mentioned earlier, all flows cause change, and depending on their character and mode of release these changes can be local or widespread. Output flows of heavy metals and persistent organic materials clearly have different potential impacts from earth that has been merely moved from one place to another. However, depending on the perspective of different individuals, both can be important. Merely summing up all flows as if they are equally important, without carefully clarifying important distinctions, can therefore be rightly criticized. Output flows were characterized according to five quality categories in Matthews (*et al.* 2000), based on nature of the flow, and whether it had been processed or not. However, as an illustration of the complexity of evaluating the potential impacts resulting from output flows, manure from animals, considered to be in the biodegradable category, can be a beneficial soil amendment or a source of considerable water pollution, depending on the quantity and the mode of the output.

A complete picture of a country's industrial metabolism, and how it has changed with time, is therefore quite difficult to portray fully. Studies that attempt to do this normally begin with data on the processed commodity flows that are ready for use or manufacture, and then use some of the techniques outlined above to arrive at estimates of the upstream hidden flows, downstream residence time in the economy, quality and ultimate fate. Studies of this kind essentially present an account of the physical activity in an economy, much the same as monetary accounts document economic activity. While both provide useful overall indicators, to obtain specific information from either of these accounting schemes it is necessary to approach them with particular questions in mind. In this manner, specific flows can be aggregated and weighed in various ways, in accordance with the preferences of the analyst. Used in this manner, national MFA accounts can be a vital complement to economic accounts.

TOTAL MATERIAL FLOWS IN THE USA

This section provides estimates of total material flows and additions to stock in the USA for the time period 1975–96. These estimates are for the domestic outputs associated with the commodity flows for food, fuels and physical goods (all other processed commodity flows), and the earth moving and dredging associated with the creation and maintenance of the built infrastructure. Table 22.2, provides information, derived from Matthews *et al.* (2000), on the total material flow for the USA. These data show that the total flow for 1996 is estimated to be 21 billion tons, or 79 tons per capita. Hidden flows constitute the

Table 22.1 List of commodities by sources and sub-groups for the USA

Renewable organic sources	Non-renewable organic sources	Metals	Minerals	Plastics*
Agriculture, fisheries and wildlife cotton, cottonseed, fishery, flax seed, fur, leather hides, mohair, raw wool, silk: raw & waste, tobacco *Paper* paper (all grades); paperboard: insulating board, hardboard, wet machine board; recycled paper *Wood* lumber, plywood, veneer; other forestry: poles and piling, fence posts, cooperage, hewn ties; other misc. products	*Primary products from petroleum & natural gas* benzene, toluene, xylenes, all other aromatics; acetylene, ethylene, propylene; butadiene and butylene fractions, 1-butene, isobutane, isobutylene, all other C4 hydrocarbons; isoprene, pentenes, mixed, piperylene, all other C5 hydrocarbons; alpha olefins, C6–C10, higher alpha olefins, dodecene, hexane, n-heptane, nonene, n-paraffins, ethane, propane, butane; all others *Asphalt & road oil* all asphalts, all road oils (grades 0 to 5)	*(Includes recycling where significant)* aluminum, antimony, arsenic, beryllium, bismuth, cadmium, cesium, chromium, cobalt, columbium, gallium, germanium, gold, indium, iron & steel, lead, magnesium, manganese, mercury, molybdenum, nickel, platinum group, rare earth, rhenium, selenium, silicon, silver, tantalum, tellurium, thallium, tin, titanium metal, tungsten, vanadium, zinc	*Construction materials* crushed stone, dimension stone, sand & gravel *Industrial minerals* abrasives, asbestos, barite, bauxite (refractory), bromine, calcium, cement, clays, diamond, diatomite, feldspar, fluorspar, graphite, natural, gypsum, hafnium, helium, industrial sand & gravel, iodine, iron oxide pigments, kyanite, lime (stone), lithium, magnesium compounds, mica, nitrogen (ammonia), peat, perlite, crude, phosphate (P2O5), potash (K2O), pumice & pumicite, quartz crystal, salt, soda ash,	alkyd-acrylate copolymer, phthalic anhydride type, polybasic acid type, all other alkyd resins; epoxy resins, phenolic and tar acid resins, melamine-formaldehyde resin, polyester resins, unsaturated, polyether and polyester polyols, polyurethanes, other thermosetting resins; polymethyl methacrylates (PMMA), other acrylic resins; engineering plastics, polyamide resins, polyethylene terephthalate (PET), all other saturated polyesters; ethylene-vinyl acetate and related, low-density polyethylenes (LDPE), high-density

	Lubricants all lubricating oils, lubricants in greases *Misc. oils & waxes* petrolatum, absorption oil, all waxes, all other non-fuel oils *Other products* natural gas for carbon black, coal for chemical use, petroleum coke		sodium sulfate, strontium, sulfur, talc & pyrophylite, thorium, titanium Tio2, vermiculite, zircon	polyethylenes (HDPE), polypropylenes, polystyrenes, all other styrene plastics, polyvinyl acetate, polyvinyl chloride (PVC), other vinyl resins, other thermoplastic resins

Note: * Plastics are derived from the primary products from petroleum and natural gas.

Sources: US Department of Agriculture, *Agricultural Statistics Yearbook*; Ulrich (1990); Howard (1997); US Department of Commerce, *Fisheries of the United States* and *Statistical Abstract of USA*; US International Trade Commission, *Synthetic Organic Chemicals*; US Department of Energy, *Annual Energy Review*; US Geological Survey, *Minerals Yearbook* and *Mineral Commodity Summaries*; Manthy (1978); *Modern Plastics,* January editions.

Table 22.2 Hidden and processed material flows in the USA, 1975–96 (thousand metric tons)

	Intensity of use			Hidden flows				
Year	Population (millions)	Total material flow including additions to stock	Total MFA/ capita (MT)	Total hidden flows	Hidden flows per capita (MT)	Minerals, mining overburden and waste	Coal mining overburden and waste	Earth moving for infra-structure creation
1975	215 973	20 896 245	97	17 192 354	80	1 394 271	5 043 965	3 960 248
1976	218 035	21 503 343	99	17 539 492	80	1 442 879	5 268 554	4 110 773
1977	220 239	21 738 337	99	17 662 095	80	1 310 446	5 847 632	3 805 110
1978	222 585	21 375 618	96	17 093 056	77	1 475 475	5 729 793	3 241 419
1979	225 055	21 771 043	97	17 450 954	78	1 567 057	5 683 300	3 533 743
1980	227 726	21 342 089	94	17 385 206	76	1 416 361	5 926 558	3 488 840
1981	229 966	21 259 869	92	17 520 339	76	1 521 250	5 989 966	3 616 521
1982	232 188	20 316 096	87	16 857 076	73	1 027 724	5 856 084	3 448 536
1983	234 307	19 369 742	83	15 759 806	67	1 121 400	5 171 570	3 385 289
1984	236 348	20 670 609	87	16 786 692	71	1 195 798	5 853 806	3 569 824
1985	238 466	19 866 626	83	15 910 969	67	1 216 504	5 402 715	3 322 865
1986	240 651	19 967 686	83	15 928 483	66	1 173 242	5 592 395	3 432 597
1987	242 804	20 024 004	82	15 780 427	65	1 304 011	5 664 322	3 220 877
1988	245 021	20 204 927	82	15 824 197	65	1 721 256	5 863 572	2 913 839
1989	247 342	20 902 772	85	16 616 678	67	1 988 734	5 947 665	3 317 126
1990	249 913	20 954 175	84	16 765 680	67	2 235 625	6 029 096	3 318 473
1991	252 650	20 129 914	80	16 176 599	64	2 299 747	5 756 733	3 087 425
1992	255 419	20 709 427	81	16 504 327	65	2 365 599	5 763 316	3 329 361
1993	258 137	20 066 911	78	15 727 375	61	2 316 250	5 673 111	2 966 729
1994	260 660	20 639 878	79	16 050 117	62	2 393 716	5 910 457	2 853 623
1995	263 034	20 530 095	78	15 904 228	60	2 463 852	5 878 950	2 894 809
1996	265 455	21 078 209	79	16 332 950	62	2 478 403	6 006 355	3 105 838

bulk of the flows, with processed material that enters use accounting for only 17–21 per cent of the total. During the period studied, hidden flows declined overall owing to decreases in erosion and earth moving for infrastructure, associated with increased soil conservation efforts and the completion of the interstate highway program. This decline was somewhat offset by increases in the hidden flows associated with minerals, and coal mining overburden and waste. The estimates of hidden flows provided by Adriaanse *et al.* (1997) and Matthews *et al.* (2000) are the most comprehensive known to exist. The reader is referred to these studies for the details of how these estimates were derived

Processed flows, dominated by fuels and physical goods, remained relatively constant during the period on a per capita basis. The fuels include both fossil and renewable resources, but exclude nuclear. On an energy content basis, the fuels are primarily, about 80 per cent, from non-renewable resources, coal, petroleum and natural gas (USEIA 1997). Processed agricultural flows represent the food for both humans and livestock. Flows of water and air are not included in the data shown.

Table 22.2 (cont.)

	Hidden flows			Processed flows				
Year	Dredging	Erosion	Other	Total processed flows	Processed flows per capita (MT)	Fuels all types	Physical goods	Agricultural flows
1975	559 625	5 525 300	708 944	3 703 891	17	1 599 079	1 943 436	161 376
1976	517 000	5 472 562	727 725	3 963 850	18	1 700 843	2 098 231	164 776
1977	511 500	5 420 327	767 080	4 076 242	19	1 719 273	2 196 268	160 701
1978	489 500	5 368 591	788 278	4 282 562	19	1 749 521	2 376 216	156 825
1979	490 875	5 317 348	858 631	4 320 089	19	1 772 791	2 391 248	156 051
1980	511 500	5 266 595	775 353	3 956 883	17	1 715 818	2 082 858	158 206
1981	596 750	5 216 326	579 526	3 739 530	16	1 659 092	1 923 001	157 437
1982	477 125	5 166 536	881 071	3 459 020	15	1 592 609	1 706 951	159 460
1983	497 750	4 952 406	631 391	3 609 936	15	1 594 104	1 855 502	160 330
1984	578 875	4 747 151	841 239	3 883 918	16	1 654 575	2 074 852	154 490
1985	519 750	4 550 402	898 734	3 955 657	17	1 657 422	2 140 755	157 481
1986	536 250	4 361 808	832 192	4 039 204	17	1 698 614	2 182 311	158 279
1987	458 150	4 338 900	794 168	4 243 577	17	1 766 066	2 321 626	155 885
1988	496 238	4 172 384	656 909	4 380 729	18	1 859 120	2 362 282	159 328
1989	562 238	4 012 258	788 657	4 286 094	17	1 887 318	2 240 556	158 220
1990	478 500	3 858 278	845 708	4 188 495	17	1 802 384	2 224 518	161 592
1991	515 625	3 710 207	806 862	3 953 315	16	1 829 323	1 961 225	162 766
1992	438 763	3 684 173	923 115	4 205 100	16	1 857 874	2 182 727	164 499
1993	473 000	3 542 784	755 502	4 339 536	17	1 871 066	2 302 515	165 955
1994	517 963	3 406 820	967 538	4 589 761	18	2 020 434	2 397 834	171 493
1995	448 388	3 406 820	811 409	4 625 867	18	2 060 452	2 391 788	173 627
1996	458 700	3 406 820	876 834	4 745 259	18	2 093 051	2 486 667	165 540

HISTORICAL USE OF MATERIAL FOR PHYSICAL GOODS IN THE USA

The use of minerals and metals for physical goods in the USA has been documented by the US Geological Survey (1900–23, 1996–present) and the US Bureau of Mines (1924–95) in their *Minerals Yearbooks*. Beginning around 1990, the area of concern was expanded to include physical goods produced from all sources, including forest, agricultural and non-renewable organic resources. Owing to the continuity and level of detail with which these data were compiled, reliable information is available for the entire 20th century. An overview of these data, disaggregated by material source, is presented in Figure 22.2 and again in Figure 22.3 in a semi-logarithmic format. (See Table 22A.1 for the data underlying these figures.) The quantities shown are the annual apparent inputs to the use phase (domestic production + imports + recycling − exports) of each processed commodity flow aggregated by material source category. The quantities of material

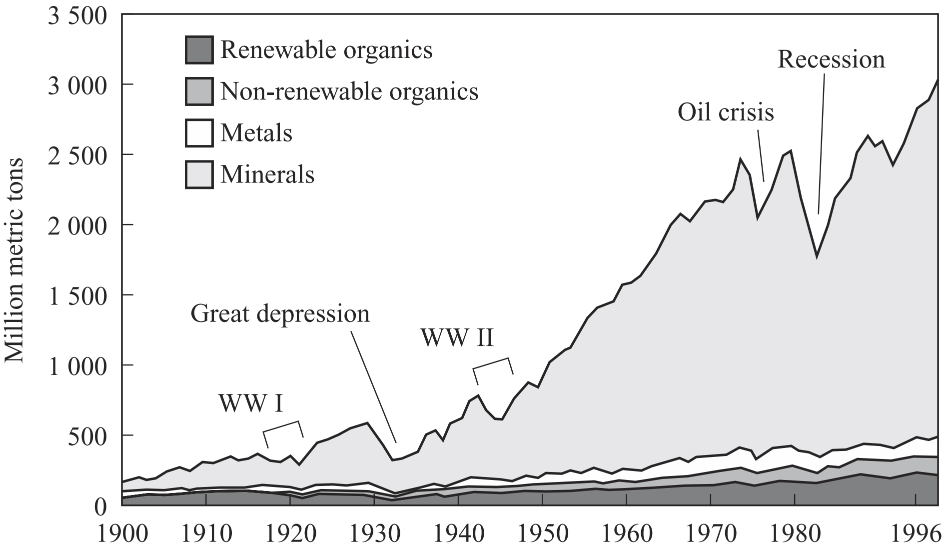


Figure 22.2 Processed flows for physical goods in the USA, 1900–96

embedded in imports and exports of finished goods have not been considered. For some processed flows these can be significant but, overall, for the USA they are small in relation to total flows (Matos and Wagner 1998).

It may be noted that the total for the processed physical goods presented in this section is somewhat higher than that presented in the earlier section: 2957 million versus 2487 million. This arises from the fact that more commodities were included here, the previous data were for outputs and additions to stock, resulting in deductions having been made for recycled flows, and in some cases different data sources were used.

On the basis of these figures a number of observations can be made. Overall, during the 20th century processed flows for physical goods in the USA rose exponentially, at a rate much faster than population, with fluctuations during business cycles, until about 1970, at which time a temporary leveling off occurred. During the last two decades of the 20th century, overall flows for physical goods once again appeared to rise faster than population. During the century, the annual use of physical goods in the USA increased about fivefold, from two to 11 tons on a per capita basis, and increasing amounts of material were obtained from non-renewable resources, dominated by the minerals category which includes the massive flows of crushed stone, sand and gravel. While flows from each source increased during the century, the rates of increase differed markedly.

Table 22.3. provides a snapshot of per capita use at the beginning and the end of the century. Increases occurred in all but one resource category, and there are considerable differences in what was used then and what is used now.

Within the minerals and metals categories (Figure 22.4 – underlying data in Table 22A.2), the rate of growth in construction minerals flattened after the completion of the interstate highway program in the 1970s, dipped in response to the recession in the early 1980s, and currently appears to be rising again. Metals use, dominated by steel, generally grew during the century but leveled off, declining slightly, from 1975 until about 1991.

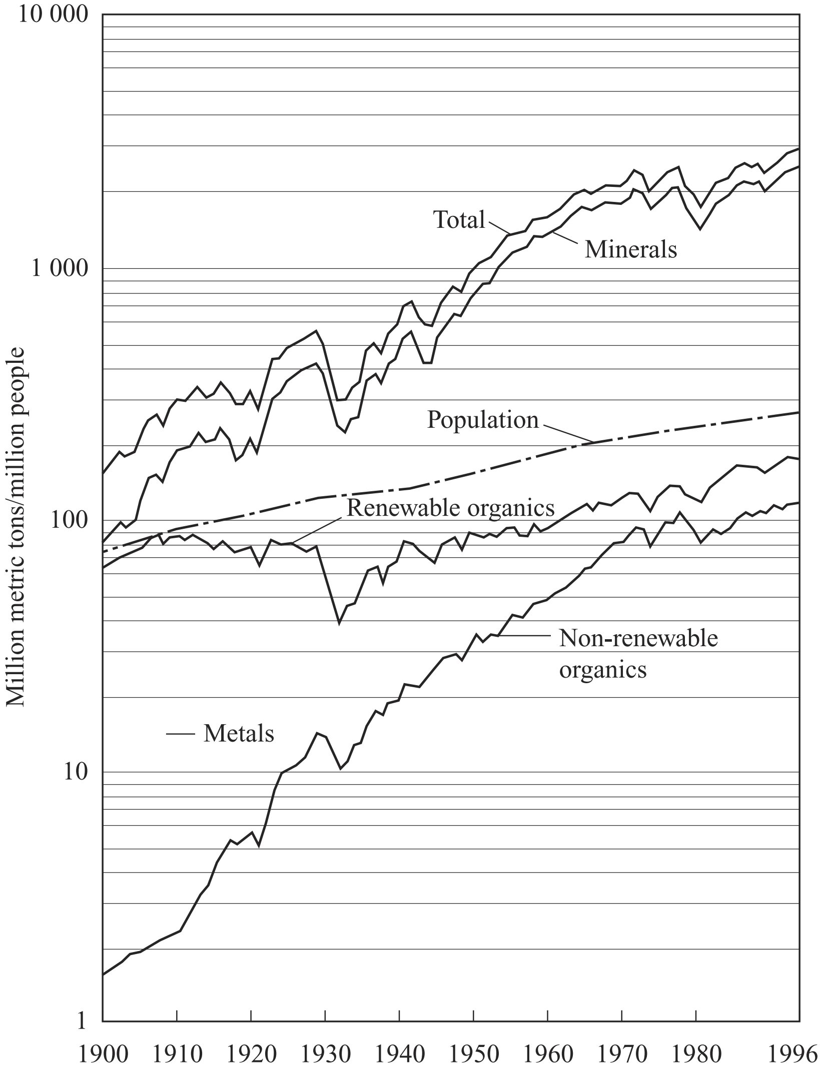


Figure 22.3 Processed flows for physical goods in the USA, 1900–96 (log scale)

Table 22.3 Sources of physical goods in the USA

Source	Per capita use (metric tons)	
	1900	1996
All sources	2.11	11.14
Minerals	1.10	9.52
Metals	0.12	0.53
Non-renewable organics	0.02	0.44
Renewable organics	0.87	0.65

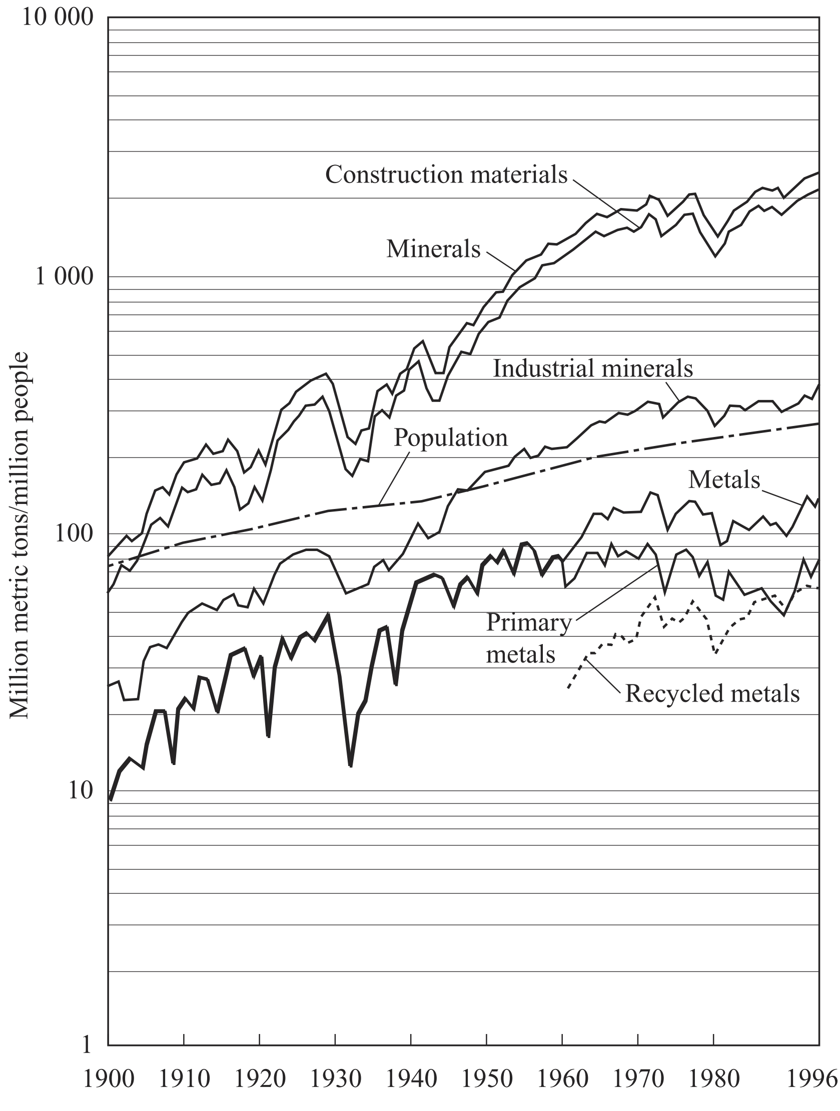


Figure 22.4 Physical goods derived from metals and minerals in the USA, 1900–96

Interestingly, the use of metals from secondary sources, recycling, was in 1996 roughly equal to that from primary resources However, in the last several years this encouraging trend seems to be reversing. It should be noted that all metals do not necessarily exhibit the same trends as shown here. Industrial minerals rose rapidly during the first half of the century, but followed the population trend from about 1950 onward. Currently, there is some recycling of construction minerals, estimated to be about 100 million MT per year, of the order of 10 per cent of annual input, but the data are poor and no quantities are shown in the data presented here.

During the 20th century, the extensive per capita use of renewable organic material

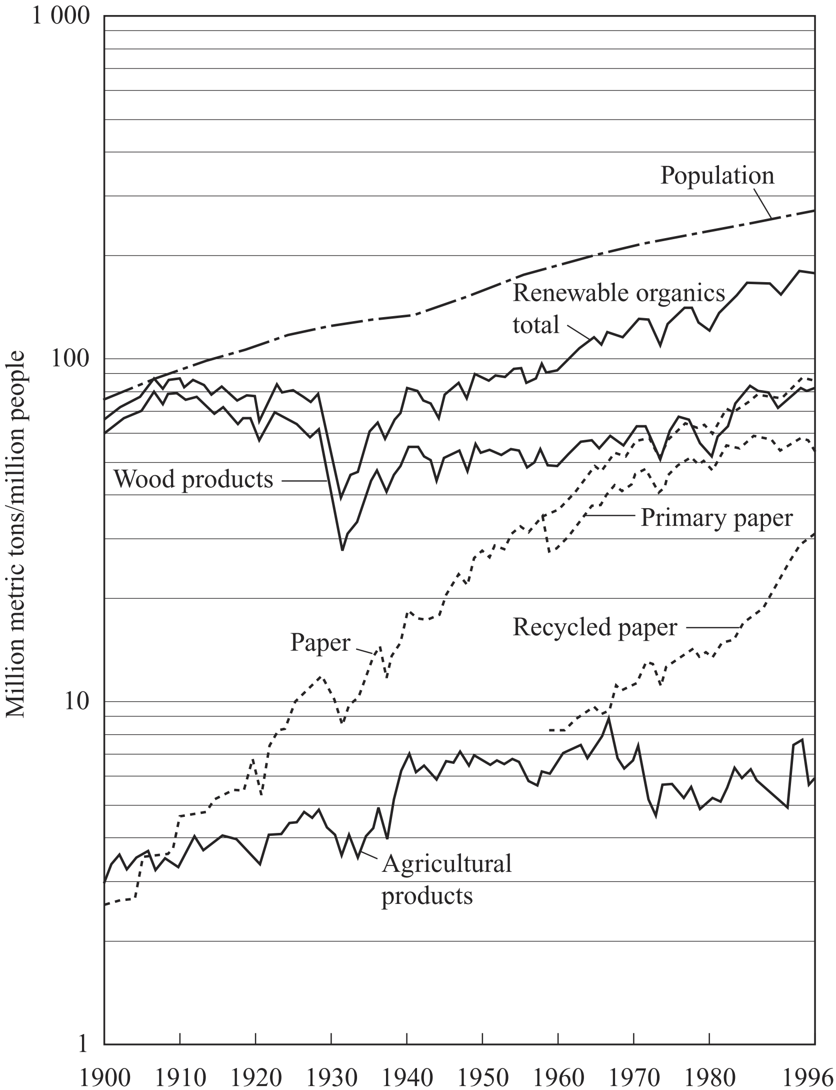


Figure 22.5 Physical goods derived from renewable organic forest and agricultural sources in the USA, 1900–96

(forest and agricultural resources) declined until after World War II, when it began to parallel the growth This overall trend resulted from dramatically different growth rates for wood and paper. (See Figure 22.5 – underlying data in Table 22A.3). Wood use declined during the early part of the century, and then leveled off after World War II. In contrast, the growth in the use of paper was steady throughout the century. Currently, the use of wood and paper is about equal, and recycled paper is approaching 50 per cent of use. During the last couple of decades, the per capita use of wood was fairly constant, while that for paper rose. The computer and electronic age does not appear to be decreasing the

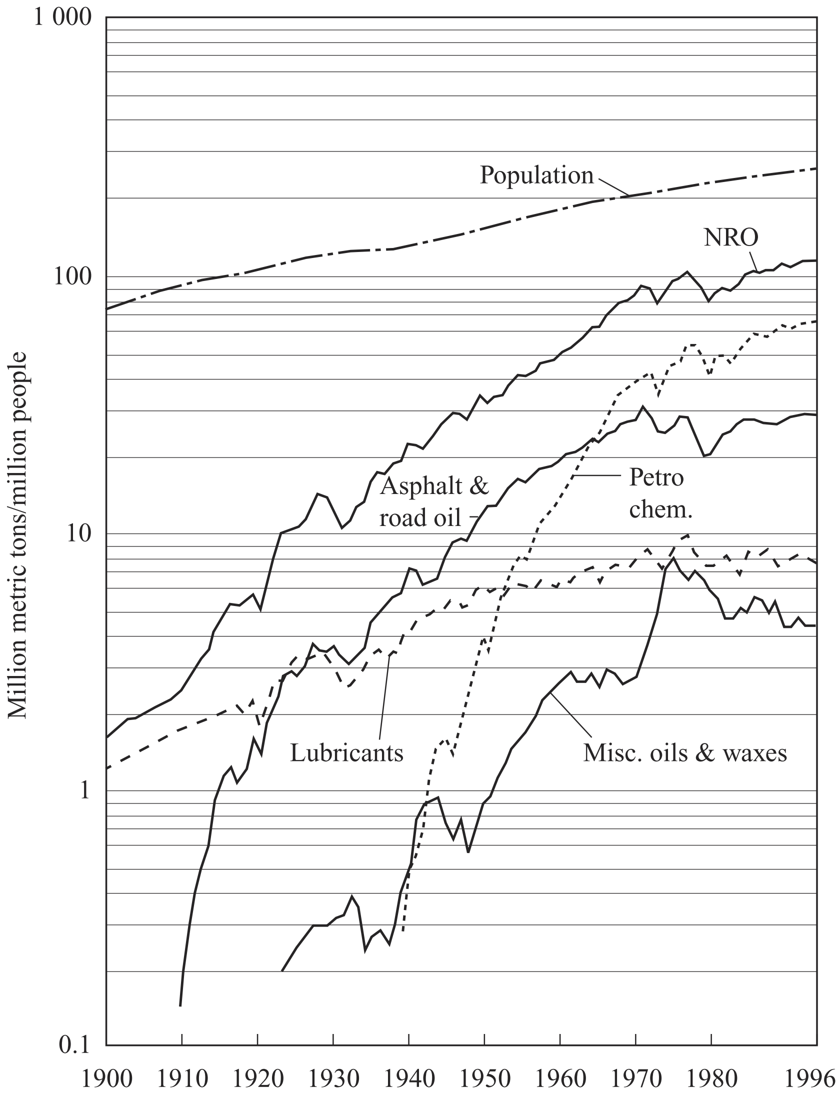


Figure 22.6 Physical goods derived from non-renewable organic sources in the USA, 1900–96

use of paper. Physical goods obtained from agricultural and fishery resources rose slightly during the century, but were an order of magnitude less than that obtained from forest resources.

The use of non-renewable organic (NRO) material (Figure 22.6 – underlying data in Table 22A.4), derived mainly from petroleum and natural gas, displayed a dramatic growth during the century. Asphalt and road oil drove the growth during the early years of the century, but from 1940, until about 1970, the use of petrochemicals rose at a rate faster than any group of commodities. In 1996, the US annual use of petrochemicals was

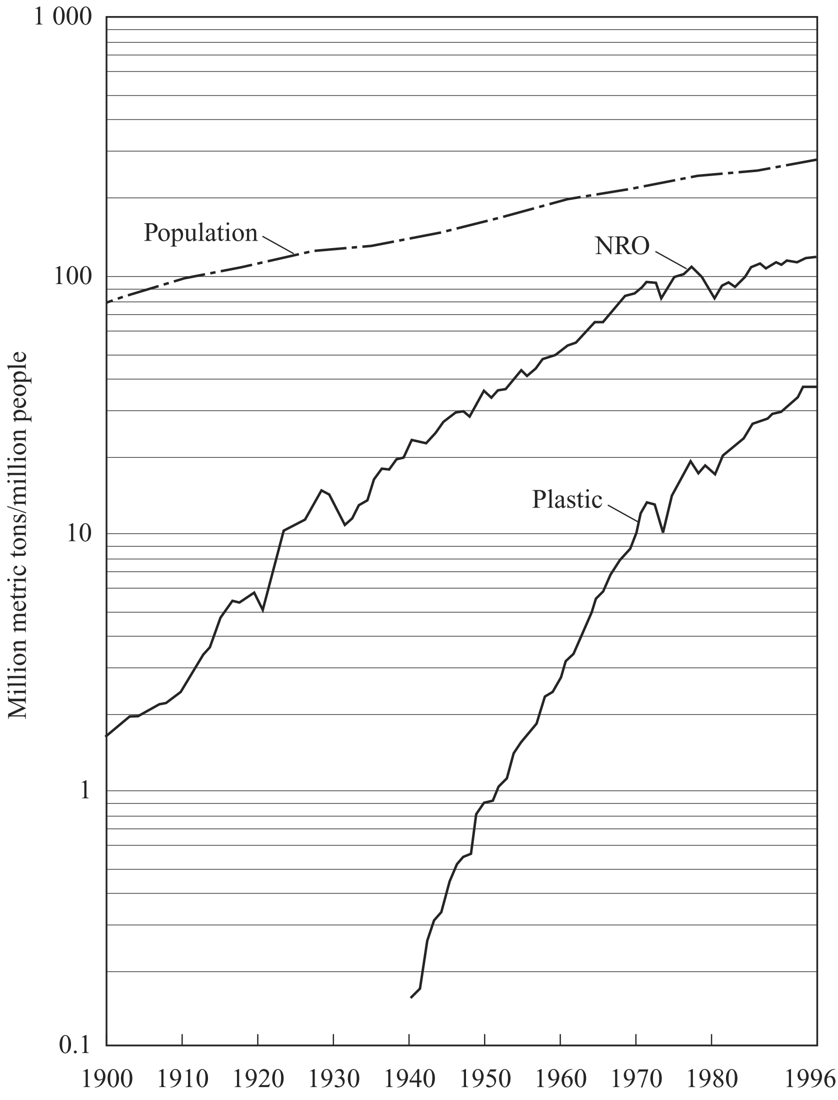


Figure 22.7 Plastic and non-renewable organic physical goods in the USA, 1900–96

about 67 million metric tons. Petrochemicals provide the feedstocks for plastic, synthetic fibers, medicinal chemicals and other materials which are now of major importance to the economy of the USA. They are also of increasing concern with respect to the impacts caused by their manufacture, use and disposal. Because of their dramatic growth and significance, plastics, derived from petrochemical material, have been portrayed separately in Figure 22.7 (underlying data in Table 22A.4). This illustrates the spectacular rise in plastic use from 1941 to the present from 0.1 to almost 40 million tons per year. The trend shows only slight signs of abating and, unfortunately, only a small amount of plastic is currently recycled.

SUMMARY: US PHYSICAL GOODS MATERIAL USE PATTERNS

During the 20th century the flow of processed physical goods to support the industrial economy of the USA increased exponentially. This trend continued after a slight pause around 1970–80. From 1975 onward, the data show that while hidden flows decreased, processed flows for fuels, physical goods and agricultural products increased at the same rate as population. With the exception of physical goods obtained from agricultural resources, the per capita use of material for this purpose continues to increase. The end of the century saw a resurgence in the use of construction materials, primary metals and wood, a consequence of a robust economy and, probably, urban sprawl. During that same period, the increased use of synthetic polymeric material affected both metals and natural fibers, and even though Americans are using increased amounts of paper, packaging applications have likely also been affected. Of major significance is the fact that they are becoming increasingly dependent on fossil fuels for material, as well as energy uses, so that disruptions in supply, or price increases, will affect multiple sectors of the US industrial system. During the 20th century, GDP generally grew considerably faster than population. Materials use kept pace with economic growth until about 1970, at which time GDP grew at a faster rate However, from an industrial ecology perspective, decoupling of material use with respect to economic growth has little meaning as long as the population continues to rise, and material use continues to grow.

GLOBAL MATERIAL USE PATTERNS

This section examines the worldwide use of processed commodity flows for physical goods and compares these with the US data. While global material flow data are not as well developed as those for the USA, some information is available permitting useful comparisons to be made. World data are for production, as distinct from use. However, this is unimportant, as long as country-specific use analyses are not attempted.

Data for five source categories (outlined in Table 22.4) of processed physical goods – minerals, metals, forestry, NRO and agriculture – were compiled for the last three decades of the 20th century. These data are provided in the Appendix as Table 22A.5. The global production of processed physical goods is shown graphically in Figure 22.8, along with world population. From this figure it can be seen that world production/use of physical goods from all five source categories has grown at about the same rate as world population, implying a relatively constant intensity of use for the period studied. Table 22.5 shows the 1996 world and US production/use per capita, for each source category.

Table 22.5 illustrates the wide discrepancy between the overall world intensity of use and that for the USA. Overall, on a per capita basis, the USA uses more than six times as much as the world average, but there are some wide differences within individual categories. These variations in specific categories may be the result of actual differences in use patterns or the result of reporting inaccuracies. It is thought that the metals and NRO comparisons may be reasonably accurate because, being highly processed materials, they are probably counted in most countries' system of economic accounts. Minerals, which are predominately construction materials, and wood products, may be used locally without formal accounting, resulting in the world production being understated. Because

Table 22.4 List of commodities, by sources and sub-groups, for the world

Agriculture	Forestry	Metals	Minerals	Non-renewable organics
castor beans	*Sawnwood*	aluminum	asbestos	asphalt (bitumen)
cotton lint	coniferous	cadmium	barite	acetone
cottonseed	non-coniferous	copper	boron	aniline
fiber, agave		lead	cement	butyl alcohol
fiber, flax	*Wood-based panels*	magnesium	clay	carbon black
fiber, hemp	fiberboard compressed	molybdenum	feldspar	ethylene glycol
fibers, other	fiberboard non-compressed	nickel	fluorspar	formaldehyde
hempseed	particle board	raw steel	granite	lubricating oils
hides, buffalo	plywood	tin	graphite	methanol
hides, cattle	veneer	vanadium	gravel	non-cellulosic contin. fiber
jute, jute-like		zinc	gypsum	non-cellulosic
linseed	*Paper and paperboards*		limestone	staple and tow
natural rubber	paper (all grades)		marble	phenol
silk	paperboard:		mica	polyethylene
sisal, etc.	insulating board		nitrogen (ammonia)	polypropylene
skins, goat	hardboard		phosphate rock	polyvinyl chloride
skins, sheep	wet machine board		salt	styrene
tobacco leaves	recovered paper		sand	synthetic rubber
wool degreased			slate	
			sulfur	
			talc	

Sources: Food & Agricultural Organization, *FAO Yearbook*; United Nations, *UN Industrial Statistics Yearbook*; United Nations Industrial Development Organization, *UNIDO Yearbook*; US Geological Survey, *Minerals Yearbook and Mineral Commodity Summaries*.

the world data shown include materials used by many advanced countries in addition to the USA, Table 22.5 understates the magnitude of the materials use gap that exists between the developed and the less developed countries. Independently of what the precise gap in equity is, if the entire world of over 6 billion people were to use 11.25 ton/capita (the US overall average of processed physical goods), worldwide production would have to increase over six and a half times, to about 67.5 billion tons per year. While the major portion of the processed flows would be construction materials and industrial minerals, the hidden flows associated with the other categories would be enormous, in terms both of quantity and of environmental impacts.

As noted earlier, on a per capita basis, the USA, and other highly developed countries, demonstrate a roughly similar overall use of processed flows. The corresponding hidden flows are also comparable. However, while in geographically large and resource-rich countries like the USA, the hidden flows mainly occur domestically, in smaller, resource-poor countries, these flows occur in the foreign countries from which imports are obtained. In a number of cases, the hidden flows associated with material imports of rich countries occur in countries where the domestic per capita use of processed flows per capita is quite low.

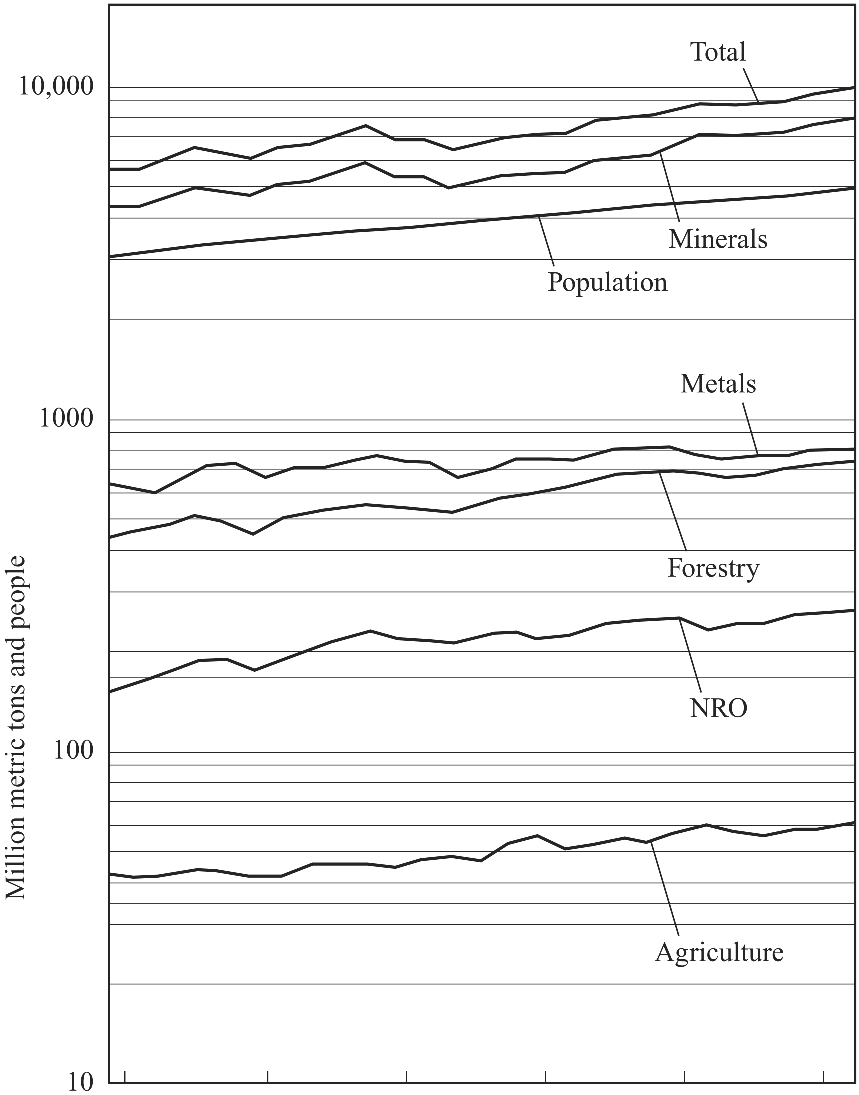


Figure 22.8 World use of materials for processed physical goods, 1970–96

Table 22.5 Global and US use of physical goods, by source category, 1996

	Total	Minerals	Metals	Forestry	Non-renewable organics	Agriculture
World (MT/capita)	1.71	1.39	0.14	0.13	0.05	0.01
USA (MT/capita)	11.14	9.52	0.53	0.63	0.44	0.02
Ratio: USA/World	6.51	6.87	3.75	4.87	9.5	2.24

CONCLUSION

All material flows cause environmental change or transformation. The material flows associated with human activity are large and, in many cases, incompatible with natural ecosystems since the flows accumulate as wastes and semi-permanent ecosystem transformations. The data presented show that, for the USA, and the rest of the world, the flows of material for processed physical goods have increased at the same rate as population, and continue to do so. For the USA, the hidden flows associated with processed commodities, and the transformations of the landscape to create the built infrastructure, are seen to be three to four times greater than the commodity flows for food, fuel and physical goods. Current economic accounts do not provide information on hidden flows.

Almost all processed goods eventually exit from the economy, some rapidly, some over much longer periods of time. Only recycling (most significant for several metals and paper), re-use or remanufacturing prevents a material input from exiting to the environment. The industrial economy of the USA is therefore essentially an open, once-through system that results in environmental impacts occurring at every stage of the material cycle. The major change necessary is to decouple the use of physical material from the output of that material to the environment. Although at present use and outputs are essentially synonymous, they need not be.

Accounts that measure the physical activity, material flows, of industrial systems are a necessary complement to national economic accounts. These accounts are required to identify trends and point to critical areas most in need of attention. Currently, the capability exists to develop overall national material flow indicators and rudimentary detailed accounts, but considerable additional work is required to develop improved and additional data and refine techniques.

APPENDIX: MATERIAL FLOW ACCOUNTS, THE USA AND THE WORLD

Table 22A.1 Processed flows for physical goods in the USA, 1900–96

Year	Renewable organics (MMT)	Non-renewable organics (MMT)	Metals (MMT)	Minerals (MMT)	All materials (MMT)	Population (millions)
1900	65.994586	1.592631	9.329899	83.749333	160.666449	76.094
1901	68.951008	1.666583	12.157966	90.236396	173.011953	77.584
1902	71.922915	1.758230	13.557046	96.808429	184.046619	79.163
1903	73.172074	1.880284	13.197156	94.563333	182.812848	80.632
1904	75.299250	1.896287	12.641886	100.628717	190.466141	82.166
1905	77.815248	1.963511	17.946839	123.631407	221.357005	83.822
1906	83.194490	2.035917	20.852993	143.126935	249.210334	85.450
1907	87.803164	2.128731	20.789491	151.207146	261.928532	87.008
1908	81.477325	2.174686	12.808727	143.017835	239.478572	88.710
1909	86.037692	2.260344	21.447471	168.995812	278.741318	90.490
1910	86.903103	2.368414	23.291039	190.080884	302.643441	92.407
1911	84.371059	2.588428	21.235902	193.914568	302.109957	93.863
1912	86.625151	2.932865	27.764103	198.925250	316.247369	95.335
1913	84.851979	3.306064	27.855580	219.798200	335.811823	97.225
1914	81.229345	3.556632	20.327008	206.048507	311.161491	99.111
1915	78.651569	4.228431	26.490002	208.251481	317.621483	100.546
1916	82.622999	4.870854	34.148742	227.467778	349.110373	101.961
1917	78.765355	5.356453	35.880998	209.289092	329.291899	103.268
1918	74.963661	5.254777	35.985796	176.361234	292.565468	103.208
1919	76.938196	5.532662	27.957186	182.430328	292.858372	104.514
1920	77.858951	5.793154	33.878463	211.561621	329.092189	106.461
1921	67.180326	5.156864	16.467908	188.633058	277.438156	108.538
1922	75.381139	6.378621	30.563005	232.655190	344.977956	110.049
1923	82.649621	8.387128	38.832669	303.297658	433.167076	111.947
1924	79.923392	10.025612	33.025010	320.951876	443.925890	114.109
1925	80.354864	10.400140	39.465131	353.345027	483.565162	115.839
1926	80.218429	10.647863	41.866401	371.565913	504.298606	117.397
1927	76.933218	11.241949	38.970540	396.553156	523.698863	119.035
1928	75.528675	12.853045	44.385985	406.105509	538.873213	120.509
1929	78.761680	14.357696	48.545305	424.197693	565.862374	121.767
1930	65.043176	13.911425	36.516019	382.255744	497.726363	123.188
1931	50.816927	12.283693	23.598472	302.041780	388.740872	124.149
1932	40.004408	10.618158	12.717441	235.174149	298.514156	124.949
1933	45.238502	11.122928	20.537907	225.338345	302.237682	125.690
1934	46.786610	12.514756	22.823030	255.437803	337.562199	126.485
1935	53.636788	13.230332	30.135868	256.772823	353.775810	127.362
1936	62.132272	15.860730	41.973376	357.801253	477.767631	128.181
1937	66.300714	17.353095	44.734362	379.138492	507.526663	128.961
1938	57.579085	17.194813	25.726642	355.636159	456.136698	129.969
1939	65.772121	18.855915	41.413504	425.265608	551.307148	130.028
1940	70.368720	19.235215	50.868107	448.087669	588.559710	132.594

Table 22A.1 (cont.)

Year	Renewable organics (MMT)	Non-renewable organics (MMT)	Metals (MMT)	Minerals (MMT)	All materials (MMT)	Population (millions)
1941	82.119813	22.441839	64.892208	534.918428	704.372288	133.894
1942	80.615336	22.287218	67.285725	573.905088	744.093367	135.361
1943	76.355671	21.712777	70.088325	478.176299	646.333072	137.250
1944	74.314811	23.149280	69.700342	427.633651	594.798085	138.916
1945	68.377593	25.617120	63.542110	429.423541	586.960365	140.468
1946	78.593821	27.481167	52.537536	533.214293	691.826818	141.936
1947	82.268463	28.936400	65.656974	601.941115	778.802952	144.698
1948	85.444050	29.543454	68.415553	657.776360	841.179417	146.208
1949	76.701934	28.263107	59.973595	652.018573	816.957209	149.767
1950	89.581514	31.547167	76.549292	739.171651	936.849625	152.271
1951	88.185814	34.624521	82.528224	811.345325	1016.683885	154.878
1952	86.882820	33.147985	77.602619	860.212068	1057.845492	157.553
1953	88.788660	34.742865	88.797822	877.453220	1089.782566	160.184
1954	88.057452	35.317414	70.708765	991.944330	1186.027962	163.026
1955	93.035862	38.684785	92.094611	1079.939437	1303.754694	165.931
1956	94.988983	41.484644	91.098925	1145.333018	1372.905570	168.903
1957	86.641113	40.901202	88.316387	1175.060530	1390.919231	171.984
1958	87.455524	42.695062	69.508332	1219.933766	1419.592684	174.882
1959	96.164382	46.298914	75.887598	1311.581638	1529.932532	177.830
1960	91.012766	47.288900	82.496709	1323.622942	1544.421317	180.671
1961	91.900359	48.592171	78.275994	1361.569358	1580.337882	183.691
1962	96.255174	51.817547	86.488552	1422.093733	1656.655006	186.538
1963	101.316247	52.759041	93.752896	1496.759727	1744.587911	189.242
1964	106.839851	55.535204	104.363820	1582.130680	1848.869556	191.889
1965	110.020501	59.423697	119.113356	1678.428387	1966.985942	194.303
1966	113.644014	64.044449	119.106426	1742.126416	2038.921306	196.560
1967	110.491998	64.463034	111.851032	1694.670673	1981.476737	198.712
1968	118.484531	70.592732	125.667703	1739.554769	2054.299735	200.706
1969	117.821101	75.684560	122.686934	1801.942281	2118.134876	202.677
1970	115.149399	80.379707	120.809168	1811.966629	2128.304903	205.052
1971	121.773259	81.820738	119.404356	1797.500266	2120.498620	207.661
1972	130.046290	85.386568	125.711509	1855.617394	2196.761761	209.896
1973	130.025238	92.584263	142.632686	2051.351207	2416.593394	211.909
1974	119.723495	91.508951	138.635949	1965.856026	2315.724421	213.854
1975	108.491800	79.918668	104.027318	1708.878285	2001.316071	215.973
1976	123.365955	89.228744	116.478418	1810.024015	2139.097132	218.035
1977	131.395971	97.025733	129.068545	1906.792862	2264.283111	220.239
1978	138.040554	99.111800	133.226121	2065.951807	2436.330282	222.585
1979	138.309087	106.126973	134.850726	2101.465325	2480.752111	225.055
1980	126.866942	98.122471	115.996375	1814.613887	2155.599675	227.726
1981	123.578472	92.118992	120.810345	1628.734859	1965.242668	229.966
1982	118.014364	81.271552	90.196552	1440.175004	1729.657472	232.188
1983	132.692188	88.041085	94.922210	1573.012011	1888.667494	234.307
1984	142.377507	90.858083	112.404560	1789.702087	2135.342237	236.348

Table 22A.1 (cont.)

Year	Renewable organics (MMT)	Non-renewable organics (MMT)	Metals (MMT)	Minerals (MMT)	All materials (MMT)	Population (millions)
1985	148.740774	89.229492	108.291740	1850.117773	2196.379779	238.466
1986	156.599001	93.585871	103.131973	1933.078269	2286.395114	240.651
1987	165.283395	102.599544	108.857519	2109.130952	2485.871410	242.804
1988	164.137941	106.372856	115.540773	2191.270930	2577.322500	245.021
1989	163.246784	104.784203	109.352597	2136.808411	2514.191994	247.342
1990	161.530171	107.111482	109.849078	2165.999001	2544.489731	249.913
1991	154.204957	107.532442	99.309694	2020.242372	2381.289465	252.650
1992	160.036276	112.553363	107.072081	2103.308241	2482.969961	255.419
1993	168.401815	110.219823	122.209883	2228.040060	2628.871581	258.137
1994	175.891263	114.363319	139.166242	2358.874967	2788.295791	260.660
1995	173.457517	114.865787	129.271909	2409.784192	2827.379405	263.034
1996	174.182348	115.578125	139.277507	2528.583467	2957.621447	265.455

Table 22A.2 Physical goods derived from metals and minerals in the USA, 1900–96 (MMT)

Year	Primary metals	Recycled metals	Metals total	Industrial minerals	Construction materials	Minerals total
1900	9.329899		9.329899	25.373832	58.375502	83.749333
1901	12.157966		12.157966	25.995581	64.240815	90.236396
1902	13.557046		13.557046	22.434429	74.374000	96.808429
1903	13.197156		13.197156	22.332757	72.230576	94.563333
1904	12.641886		12.641886	22.705264	77.923454	100.628717
1905	17.946839		17.946839	32.931407	90.700000	123.631407
1906	20.852993		20.852993	36.100935	107.026000	143.126935
1907	20.789491		20.789491	36.925146	114.282000	151.207146
1908	12.808727		12.808727	35.991835	107.026000	143.017835
1909	21.447471		21.447471	40.201812	128.794000	168.995812
1910	23.291039		23.291039	43.146884	146.934000	190.080884
1911	21.235902		21.235902	48.794568	145.120000	193.914568
1912	27.764103		27.764103	50.177250	148.748000	198.925250
1913	27.855580		27.855580	52.910200	166.888000	219.798200
1914	20.327008		20.327008	51.858507	154.190000	206.048507
1915	26.490002		26.490002	51.340481	156.911000	208.251481
1916	34.148742		34.148742	55.137778	172.330000	227.467778
1917	35.880998		35.880998	56.913092	152.376000	209.289092
1918	35.985796		35.985796	52.102234	124.259000	176.361234
1919	27.957186		27.957186	51.822328	130.608000	182.430328
1920	33.878463		33.878463	59.185621	152.376000	211.561621
1921	16.467908		16.467908	53.490058	135.143000	188.633058
1922	30.563005		30.563005	66.674190	165.981000	232.655190
1923	38.832669		38.832669	74.795195	228.502463	303.297658
1924	33.025010		33.025010	77.667209	243.284667	320.951876
1925	39.465131		39.465131	83.057691	270.287336	353.345027

Table 22A.2 (cont.)

Year	Primary metals	Recycled metals	Metals total	Industrial minerals	Construction materials	Minerals total
1926	41.866401		41.866401	83.955471	287.610442	371.565913
1927	38.970540		38.970540	85.040134	311.513022	396.553156
1928	44.385985		44.385985	86.733695	319.371813	406.105509
1929	48.545305		48.545305	86.274975	337.922718	424.197693
1930	36.516019		36.516019	81.991829	300.263915	382.255744
1931	23.598472		23.598472	68.508288	233.533493	302.041780
1932	12.717441		12.717441	58.274776	176.899373	235.174149
1933	20.537907		20.537907	59.579244	165.759102	225.338345
1934	22.823030		22.823030	61.692001	193.745802	255.437803
1935	30.135868		30.135868	63.650825	193.121998	256.772823
1936	41.973376		41.973376	75.125633	282.675620	357.801253
1937	44.734362		44.734362	78.322489	300.816004	379.138492
1938	25.726642		25.726642	71.454708	284.181451	355.636159
1939	41.413504		41.413504	78.784352	346.481256	425.265608
1940	50.868107		50.868107	83.194313	364.893356	448.087669
1941	64.892208		64.892208	97.095923	437.822505	534.918428
1942	67.285725		67.285725	108.300266	465.604822	573.905088
1943	70.088325		70.088325	98.085251	380.091048	478.176299
1944	69.700342		69.700342	97.682470	329.951181	427.633651
1945	63.542110		63.542110	101.521273	327.902268	429.423541
1946	52.537536		52.537536	128.573476	404.640817	533.214293
1947	65.656974		65.656974	139.680402	462.260713	601.941115
1948	68.415553		68.415553	149.615098	508.161262	657.776360
1949	59.973595		59.973595	146.725245	505.293328	652.018573
1950	76.549292		76.549292	160.092966	579.078685	739.171651
1951	82.528224		82.528224	172.242287	639.103038	811.345325
1952	77.602619		77.602619	176.676914	683.535154	860.212068
1953	88.797822		88.797822	182.329327	695.123893	877.453220
1954	70.708765		70.708765	184.227271	807.717059	991.944330
1955	92.094611		92.094611	200.068714	879.870723	1079.939437
1956	91.098925		91.098925	208.860053	936.472965	1145.333018
1957	88.316387		88.316387	197.990245	977.070285	1175.060530
1958	69.508332		69.508332	198.200080	1021.733686	1219.933766
1959	75.887598		75.887598	215.739703	1095.841935	1311.581638
1960	82.496709		82.496709	213.224564	1110.398378	1323.622942
1961	78.275994		78.275994	213.503270	1148.066088	1361.569358
1962	62.597656	23.890896	86.488552	216.484219	1205.609514	1422.093733
1963	66.108771	27.644125	93.752896	226.966673	1269.793054	1496.759727
1964	74.437769	29.926051	104.363820	242.142363	1339.988317	1582.130680
1965	85.456697	33.656659	119.113356	256.436396	1421.991991	1678.428387
1966	84.636416	34.470010	119.106426	269.214967	1472.911449	1742.126416
1967	74.371691	37.479341	111.851032	269.582569	1425.088104	1694.670673
1968	88.229340	37.438363	125.667703	280.447058	1459.107711	1739.554769
1969	81.803560	40.883374	122.686934	292.547166	1509.395115	1801.942281
1970	83.639156	37.170012	120.809168	288.610244	1523.356385	1811.966629

Table 22A.2 (cont.)

Year	Primary metals	Recycled metals	Metals total	Industrial minerals	Construction materials	Minerals total
1971	81.455505	37.948851	119.404356	293.550053	1503.950213	1797.500266
1972	80.096614	45.614895	125.711509	310.855809	1544.761585	1855.617394
1973	90.699439	51.933247	142.632686	325.588702	1725.762505	2051.351207
1974	82.753825	55.882124	138.635949	320.423790	1645.432236	1965.856026
1975	60.897400	43.129918	104.027318	281.130584	1427.747701	1708.878285
1976	70.475480	46.002938	116.478418	302.383219	1507.640796	1810.024015
1977	83.531066	45.537479	129.068545	318.340180	1588.452682	1906.792862
1978	86.023691	47.202430	133.226121	337.506396	1728.445411	2065.951807
1979	80.756807	54.093919	134.850726	344.596348	1756.868977	2101.465325
1980	67.425772	48.570603	115.996375	317.276611	1497.337276	1814.613887
1981	75.319653	45.490692	120.810345	296.399930	1332.334929	1628.734859
1982	57.089654	33.106898	90.196552	258.256955	1181.918049	1440.175004
1983	55.496100	39.426110	94.922210	278.774710	1294.237301	1573.012011
1984	69.457291	42.947269	112.404560	313.339186	1476.362901	1789.702087
1985	62.225796	46.065944	108.291740	311.562789	1538.554984	1850.117773
1986	56.347480	46.784493	103.131973	301.312709	1631.765560	1933.078269
1987	57.088171	51.769348	108.857519	308.135518	1800.995434	2109.130952
1988	59.387326	56.153447	115.540773	322.776556	1868.494374	2191.270930
1989	54.752401	54.600196	109.352597	321.004388	1815.804023	2136.808411
1990	52.090742	57.758336	109.849078	323.879029	1842.119972	2165.999001
1991	47.660339	51.649355	99.309694	301.267372	1718.975000	2020.242372
1992	53.687378	53.384703	107.072081	306.732241	1796.576000	2103.308241
1993	64.836349	57.373534	122.209883	315.238060	1912.802000	2228.040060
1994	77.905578	61.260664	139.166242	335.474967	2023.400000	2358.874967
1995	66.960211	62.311698	129.271909	332.384192	2077.400000	2409.784192
1996	78.851097	60.426410	139.277507	375.183467	2153.400000	2528.583467

Table 22A.3 Physical goods derived from renewable organic forest and agricultural sources in the USA, 1900–96 (MMT)

Year	Agricultural products	Wood products	Primary paper	Recycled paper	Paper total
1900	3.035731	62.958855			2.639048
1901	3.457804	65.493204			2.665705
1902	3.663608	68.259308			2.692632
1903	3.304247	69.867828			2.719830
1904	3.563603	71.735648			2.747303
1905	3.677537	74.137711			3.574782
1906	3.743508	79.450982			3.610891
1907	3.310555	84.492609			3.647365
1908	3.573201	77.904124			3.684207
1909	3.494946	82.542746			3.721421
1910	3.377123	83.525980			4.700451
1911	3.708976	80.662083			4.747930

Table 22A.3 (cont.)

Year	Agricultural products	Wood products	Primary paper	Recycled paper	Paper total
1912	4.087806	82.537345			4.795889
1913	3.804826	81.047153			4.844332
1914	3.888815	77.340529			4.893265
1915	3.999737	74.651833			5.381708
1916	4.138344	78.484655			5.436068
1917	4.120720	74.644635			5.490978
1918	4.051991	70.911669			5.691425
1919	3.856078	73.082118			5.671471
1920	3.745344	74.113607			6.902270
1921	3.517970	63.662357			5.466489
1922	4.174964	71.206176			7.133555
1923	4.216630	78.432991			8.338958
1924	4.259379	75.664013			8.417867
1925	4.557286	75.797578			9.448219
1926	4.597202	75.621227			10.506688
1927	4.860664	72.072555			10.815975
1928	4.746495	70.782180			11.293057
1929	4.934514	73.827165			12.163777
1930	4.412699	60.630477			11.173333
1931	4.282226	46.534700			10.298078
1932	3.643966	36.360442			8.828738
1933	4.178921	41.059581			9.900812
1934	3.618598	43.168012			10.239123
1935	4.066358	49.570429			11.571506
1936	4.363633	57.768639			13.288457
1937	5.011994	61.288720			14.537396
1938	4.041200	53.537884			12.282594
1939	5.280663	60.491458			14.465743
1940	6.420806	63.947914			15.198599
1941	7.275560	74.844253			18.521847
1942	6.409062	74.206274			17.940460
1943	6.640111	69.715559			17.629359
1944	6.262246	68.052566			17.636615
1945	6.066811	62.310782			17.836155
1946	6.796448	71.797373			20.416570
1947	6.795505	75.472958			22.447343
1948	7.305200	78.138850			23.657281
1949	6.627716	70.074218			22.398365
1950	7.101155	82.480359			26.313884
1951	6.860592	81.325223			27.718827
1952	6.714987	80.167833			26.318419
1953	6.818081	81.970579			28.443520
1954	6.684437	81.373015			28.460753
1955	6.877092	86.158770			31.490133
1956	6.834570	88.154412			33.101872

Table 22A.3 (cont.)

Year	Agricultural products	Wood products	Primary paper	Recycled paper	Paper total
1957	5.991746	80.649367			31.988076
1958	5.801985	81.653539			31.852933
1959	6.392491	89.771891			35.123575
1960	6.375154	49.068700	27.170092	8.398820	35.568912
1961	6.743036	48.524500	28.234910	8.397913	36.632823
1962	7.223147	50.610600	29.958210	8.463217	38.421427
1963	7.403653	54.238600	30.708299	8.965695	39.673994
1964	7.647610	56.959600	33.042917	9.189724	42.232641
1965	7.058768	58.320100	35.084574	9.557059	44.641633
1966	7.541340	58.229400	37.996951	9.876323	47.873274
1967	8.054511	55.236300	37.919856	9.281331	47.201187
1968	9.125727	58.773600	40.981888	9.603316	50.585204
1969	6.890466	57.413100	42.309736	11.207799	53.517535
1970	6.529800	55.961900	41.573252	11.084447	52.657699
1971	6.790148	60.859700	42.742375	11.381036	54.123411
1972	7.593127	63.943500	46.381259	12.128404	58.509663
1973	5.420671	63.943500	47.425216	13.235851	60.661067
1974	4.834712	56.234000	45.529586	13.125197	58.654783
1975	5.769515	51.971100	39.610504	11.140681	50.751185
1976	5.861384	59.499200	45.078807	12.926564	58.005371
1977	5.660375	64.669100	47.526800	13.539696	61.066496
1978	5.408130	68.478500	50.311290	13.842634	64.153924
1979	5.727455	66.845900	51.341642	14.394090	65.735732
1980	5.073168	58.229400	49.601109	13.963265	63.564374
1981	5.142412	54.238600	50.123541	14.073919	64.197460
1982	5.408500	51.789700	47.283724	13.532440	60.816164
1983	5.323085	60.406200	52.348412	14.614491	66.962903
1984	5.744306	64.669100	56.378213	15.585888	71.964101
1985	6.586664	70.927400	55.898410	15.328300	71.226710
1986	6.168516	78.092700	55.532889	16.804896	72.337785
1987	6.484021	83.081200	58.165910	17.552264	75.718174
1988	5.940815	80.450900	59.254310	18.491916	77.746226
1989	5.725373	80.088100	58.438917	18.994394	77.433311
1990	5.529799	77.276400	58.123281	20.600691	78.723972
1991	5.301860	71.743700	54.722031	22.437366	77.159397
1992	5.022100	74.950565	55.316116	24.747495	80.063611
1993	7.571519	77.805330	56.549636	26.475330	83.024966
1994	7.888144	81.171474	57.940067	28.891578	86.816226
1995	5.854348	80.499424	57.498358	29.605387	87.103745
1996	6.163590	82.316328	53.705284	31.997146	85.702430

Table 22A.4 Physical goods derived from non-renewable organic sources and plastics in the USA, 1900–96 (MMT)

Year	Primary products from petroleum and natural gas	Asphalt and road oil	Lubricants	Miscellaneous oils & waxes	Non-renewable organics	Plastics
1900		0.009000	1.200000		1.592631	
1901		0.013000	1.250000		1.666583	
1902		0.018889	1.300000		1.758230	
1903		0.041892	1.350000		1.880284	
1904		0.040275	1.400000		1.896287	
1905		0.047499	1.450000		1.963511	
1906		0.058952	1.500000		2.035917	
1907		0.125119	1.550000		2.128731	
1908		0.108674	1.600000		2.174686	
1909		0.117542	1.650000		2.260344	
1910		0.146197	1.700000		2.368414	
1911		0.251413	1.750000		2.588428	
1912		0.375810	1.800000		2.932865	
1913		0.499778	1.850000		3.306064	
1914		0.611744	1.900000		3.556632	
1915		0.954909	1.950000		4.228431	
1916		1.143474	2.000000		4.870854	
1917		1.222112	2.050000	0.100000	5.356453	
1918		1.090595	2.078496	0.109921	5.254777	
1919		1.169638	2.045113	0.115271	5.532662	
1920		1.583871	2.216842	0.133953	5.793154	
1921		1.389808	1.811579	0.148558	5.156864	
1922		1.856908	2.328571	0.156609	6.378621	
1923		2.153559	2.649173	0.171897	8.387128	
1924		2.792989	2.725414	0.200602	10.025612	
1925		2.882782	3.094887	0.225198	10.400140	
1926		2.806719	3.393684	0.242650	10.647863	
1927		3.113285	3.258496	0.271114	11.241949	
1928		3.688178	3.483910	0.294466	12.853045	
1929		3.518930	3.550226	0.296084	14.357696	
1930		3.497249	3.246466	0.295632	13.911425	
1931		3.612205	2.996090	0.318369	12.283693	
1932		3.300000	2.498346	0.325780	10.618158	
1933		3.112034	2.579248	0.378944	11.122928	
1934		3.352627	2.779549	0.349662	12.514756	
1935		3.564243	2.956541	0.239972	13.230332	
1936		4.596963	3.356842	0.274961	15.860730	
1937		4.937820	3.507218	0.288097	17.353095	
1938		5.277521	3.192932	0.249946	17.194813	
1939		5.762601	3.565865	0.299409	18.855915	
1940	0.270000	5.942683	3.712782	0.413286	19.235215	
1941	0.500000	7.332905	4.549624	0.493956	22.441839	0.157
1942	0.550000	7.291963	4.369474	0.769784	22.287218	0.169

Table 22A.4 (cont.)

Year	Primary products from petroleum and natural gas	Asphalt and road oil	Lubricants	Miscellaneous oils & waxes	Non-renewable organics	Plastics
1943	0.710510	6.410891	4.730677	0.886967	21.712777	0.260
1944	1.273016	6.547855	4.851579	0.922186	23.149280	0.308
1945	1.502740	6.740924	5.313383	0.934731	25.617120	0.336
1946	1.599442	8.115512	5.246767	0.745143	27.481167	0.427
1947	1.422836	9.292904	5.485865	0.652289	28.936400	0.501
1948	1.776956	9.554455	5.410977	0.763192	29.543454	0.549
1949	2.264098	9.452145	4.977594	0.572013	28.263107	0.557
1950	2.975062	10.80858	15.842556	0.725472	31.547167	0.828
1951	3.907578	11.924092	6.359699	0.881560	34.624521	0.894
1952	3.571618	12.863036	5.739098	0.944674	33.147985	0.903
1953	5.060738	13.004950	6.089774	1.128798	34.742865	1.047
1954	6.104484	13.828383	5.795038	1.247946	35.317414	1.103
1955	7.345266	15.287459	6.387519	1.448599	38.684785	1.421
1956	8.125692	16.408086	6.606466	1.579201	41.484644	1.530
1957	8.214676	15.857426	6.197744	1.692856	40.901202	1.673
1958	9.489962	16.880363	5.935639	1.883616	42.695062	1.791
1959	11.087588	17.927888	6.447820	2.234627	46.298914	2.283
1960	11.870738	18.250825	6.417444	2.364730	47.288900	2.361
1961	12.967602	18.729373	6.245113	2.568343	48.592171	2.645
1962	14.963840	19.983498	6.556391	2.710459	51.817547	3.142
1963	15.944934	20.495050	6.556391	2.851379	52.759041	3.319
1964	18.097348	20.907591	6.885414	2.676530	55.535204	3.854
1965	20.207540	22.128713	7.085714	2.663080	59.423697	4.439
1966	22.912018	23.307261	7.360752	2.824479	64.044449	5.453
1967	24.714852	22.808251	6.635038	2.528581	64.463034	5.752
1968	28.155718	24.453300	7.288271	2.972428	70.592732	6.850
1969	32.377010	25.090264	7.335639	2.851379	75.684560	7.555
1970	35.357066	26.897690	7.472632	2.609280	80.379707	7.935
1971	36.793522	27.557756	7.416692	2.703430	81.820738	8.987
1972	39.403568	28.052805	7.969925	2.770679	85.386568	11.568
1973	41.427500	31.353135	8.872180	3.470074	92.584263	13.027
1974	42.836262	29.042904	8.526316	4.177539	91.508951	12.651
1975	35.452406	25.214521	7.518797	5.205111	79.918668	10.105
1976	41.828836	24.917492	8.421053	7.357095	89.228744	12.873
1977	45.914382	26.072607	8.721805	8.069939	97.025733	14.865
1978	46.769264	28.547855	9.473684	7.114997	99.111800	16.700
1979	54.736056	28.547855	9.924812	6.644250	106.126973	18.419
1980	54.585328	23.927393	8.285714	7.213930	98.122471	16.736
1981	49.720718	20.627063	8.000000	6.716418	92.118992	17.600
1982	42.223362	20.627063	7.285714	5.970149	81.271552	16.619
1983	49.790180	22.442244	7.571429	5.597015	88.041085	19.147
1984	49.334364	24.587459	8.142857	4.726368	90.858083	20.512
1985	47.435736	25.247525	7.571429	4.726368	89.229492	21.532
1986	51.549430	27.062706	6.714286	5.099502	93.585871	22.885

Table 22A.4 (cont.)

Year	Primary products from petroleum and natural gas	Asphalt and road oil	Lubricants	Miscellaneous oils & waxes	Non-renewable organics	Plastics
1987	55.896026	28.052805	8.428571	4.975124	102.599544	25.319
1988	59.220214	28.217822	8.000000	5.597015	106.372856	26.560
1989	58.914000	27.227723	8.285714	5.472637	104.784203	27.069
1990	59.894000	27.062706	8.571429	4.975124	107.111482	28.594
1991	62.270000	26.732673	7.571429	5.348259	107.532442	28.744
1992	64.670000	27.392739	7.714286	4.353234	112.553363	30.697
1993	62.892000	28.547855	8.000000	4.353234	110.219823	32.248
1994	65.643000	29.042904	8.285714	4.601990	114.363319	35.718
1995	66.233787	29.400000	8.140000	4.390000	114.865787	35.943
1996	66.896125	29.200000	7.850000	4.390000	115.578125	36.000

Table 22A.5 World use of materials for physical goods, 1972–96 (MMT)

Year	Agriculture	Forestry	Non-renewable organics	Metals	Minerals	Total materials
1972	42.767	480.965	173.796	658.222	4660.891	6016.641
1973	44.130	507.415	190.296	729.868	5023.812	6495.521
1974	43.678	495.415	190.907	741.819	4856.300	6328.119
1975	42.318	457.286	176.327	676.620	4749.921	6102.472
1976	42.473	504.069	188.845	712.056	5132.082	6579.525
1977	45.441	523.275	201.959	711.139	5280.790	6762.605
1978	45.900	541.587	217.103	752.929	5593.513	7151.031
1979	45.966	555.219	229.752	785.029	5983.256	7599.221
1980	44.813	552.189	220.504	756.949	5366.595	6941.050
1981	47.541	541.475	217.960	747.862	5363.043	6917.880
1982	48.529	530.831	212.870	682.488	5026.287	6501.005
1983	47.165	561.925	223.922	703.210	5247.668	6783.891
1984	53.065	592.072	231.978	753.980	5382.149	7013.245
1985	55.698	601.658	219.124	761.439	5585.947	7223.866
1986	50.959	629.912	223.722	756.353	5531.062	7192.008
1987	52.294	661.725	237.673	780.044	6054.119	7785.854
1988	54.851	687.406	248.324	828.094	6153.500	7972.175
1989	54.413	698.222	249.708	833.374	6240.414	8076.131
1990	57.521	702.597	253.360	821.413	6826.080	8660.970
1991	60.544	681.524	232.868	782.126	7190.898	8947.960
1992	57.520	676.821	242.529	770.513	7074.577	8821.960
1993	55.908	688.487	243.650	775.368	7161.568	8924.981
1994	57.423	717.677	256.813	777.715	7375.451	9185.079
1995	58.045	737.074	258.792	807.508	7681.233	9542.652
1996	58.118	748.084	266.131	806.193	7957.113	9835.638

Sources: Food and Agricultural Organization, *FAO Yearbook*; United Nations, *UN Industrial Statistics Yearbook*; United Nations Industrial Development Organization, *UNIDO Yearbook*; US Geological Survey, *Minerals Yearbook* and *Mineral Commodity Summaries*.

23. Industrial ecology: analyses for sustainable resource and materials management in Germany and Europe

Stefan Bringezu

Industrial ecology comprises the analysis of the industrial metabolism and the implementation of appropriate instruments and measures for materials and resource management on different levels. This contribution will focus on analyses for public policy support. Policy development in Germany and Europe since the mid-1990s has been influenced by material flow-related goals, such as resource efficiency factor 4 to 10 and the eco-efficiency debate. These concepts in turn have gained momentum from the availability of data on the material throughput and resource requirements of industrialized countries.

The study *Sustainable Germany* (Loske *et al.* 1996) was based on the first comprehensive material flow accounts of the German economy (Bringezu and Schütz 1995). It found widespread resonance among non-governmental organizations (NGOs) and inspired similar work by governmental agencies (UBA 1997b). The German environmental ministry prepared a draft environmental policy program (BMU 1998) which proposed concrete targets for an increase in energy and raw materials productivity. Parallel to the policy debate, the Federal Statistical Office (FSO) developed a material and energy flow information system (MEFIS) (Radermacher and Stahmer 1998).

On the European level, studies from NGOs also stimulated political action. Inspired by the study *Sustainable Netherlands* (Buitenkamp *et al.* 1992), data on selected materials consumption were provided and targets for sustainable development were proposed in studies such as *Sustainable Europe* (FoE 1995). These studies were rooted in the concept of environmental space (Opschoor 1992) and exemplified the factor 4 to 10 goal for an increase in resource efficiency (Schmidt-Bleek 1992b; Weizsäcker *et al.* 1997). These concepts have been adopted by political programs in various countries, mainly in Europe and on the level of the European Union (see Chapter 8). As a consequence the need for quantitative information on the metabolic performance of those economies was increased. This demand was reflected by the further development of material flow accounting and the provision of material flow-based indicators for sustainability by European statistical organizations (EEA 1999a, 2000; Eurostat 2000).

The official activities gained from the rapid development of material flow analysis in the 1990s (see Chapter 2). Exchange between researchers at the European and world level was fostered by the ConAccount (*www.conaccount.net*) network, described in Chapter 8. In Germany the Federal Statistical Office (FSO 1997)[1] invited institutes in the German-speaking region to join in a regular information exchange. Material flow accounts developed by the Wuppertal Institute were adopted by the FSO within the framework of integrated environmental and economic accounting (FSO 1995, 2000). The method for the derivation of indicators such as TMR (see Chapter 8) found wide international resonance following the *Resource Flows* publication (Adriaanse *et al.* 1997). This in turn stimulated the political debate at the EU level.

Historically, materials management policy started at the end of the societal throughput. Waste management first concentrated on controlled disposal. In Germany, the fundamentals of prevention and recycling were given priority in the Waste Act in 1986, which was further developed to the *Kreislaufwirtschaft* Act in 1994. Based on the initiative of the former environmental minister Klaus Töpfer, this law for the first time mandated comprehensive recycling of materials and products in the production and consumption circle. At the European level a Community Strategy for Waste Management was adopted by the European Commission in 1989. This was strengthened in a 1996 review giving priority to the recovery of material over the recovery of energy. However, most regulations of waste and environmental policy were directed towards specific problems and waste types or emissions (see EEA 1999a, where the chapter on 'Waste generation and management' provides data and information on policy framework, in particular pp. 218–20). Based on a systems perspective, the Enquête Commission of the German parliament legislated materials management from 'cradle to grave' (Enquête Commission 1994, 1998; Friege *et al.* 1998). The European Environment Agency stated that

> increasing waste quantities cannot be solved in a sustainable way by efficient waste management and recycling alone. There is an urgent need for integration of waste management into a strategy for sustainable development, where waste prevention, reduction of resource depletion and energy consumption and minimization of emissions at the source is given high priority. Waste must be analyzed and handled as an integrated part of total material flow through the society. (EEA 1999a, p. 207)

MATERIAL FLOW BALANCE FOR GERMANY

Focusing on those material flows which are linked to economic activities, a domestic material flow balance of Germany has been calculated (Schütz and Bringezu 1993; Bringezu and Schütz 1995; FSO 1995, 2000). It comprises the physical mass balance of the domestic extraction from the environment, domestic deposition and release to the environment, imports and exports (see Table 8.2). It aims to

- provide an overview of the physical basis of the economy, and combine information from different statistics (for example, production statistics and environmental statistics) in a coherent framework,
- establish a structured information base that can be used to derive indicators for progress towards sustainability, and
- develop a physical satellite that can be used for integrated economic and environmental reporting.

The overview provides the following major points of information (Table 23.1). The throughput of *water* dominates the account. This category is treated separately, because the sum of all inputs and outputs would only be meaningful in terms of water use. A distinction was made between used and unused water input. The latter comprises drainage

Table 23.1 Domestic material flow balance for Germany, 1996

Input (kg/capita)		Output (kg/capita)	
Abiotic raw materials	42 080	Waste disposal (excluding incineration)	28 463
used: minerals*	10 978	controlled waste deposition (*)	1 456
ores*	1	landfill and mine dumping*	27 007
energy carriers*	3 089		
unused: non-saleable extraction*	24 396	Soil	
excavation*	3 616	erosion (anthropogenic)	1 535
Biotic raw materials	2 748	Dissipative use of products and dissipative losses	575
plant biomass from cultivation*	2 744		
agriculture	2 411	fertilizers*	448
forestry	333	mineral fertilizers*	113
fishing/hunting*	5	organic fertilizers (dry weight)*	334
		sewage sludge (dry weight)*	13
Soil		compost (fresh weight)	11
erosion (anthropogenic)	1 535	pesticides*	0.4
		others	103
Air	13 200		
O_2 for combustion*	12 176	Emissions to air	12 283
O_2 for industrial processes	1 025	CO_2*	12 094
		NOx*, SO_2*, CO*	128
		others*	60
		Emissions of water from materials	7 814
		Emissions to water	455
		dredge excavation into North Sea	413
		N and P from sewage released to surface waters	4
		other substances from sewage released to surface waters	38
Total	59 563	Total	51 124
Imports*	5 805	Exports*	2 785
Total	65 369	Total	53 909
		Balance: material added to technosphere	11 460
Water	597 782	Waste water	597 782
used*	533 330	treated*	72 869
unused*	64 448	untreated*	425 065
water imports*	4	water losses and evaporation*	35 298
		water diversion*	64 448
		water exports*	101

Note: * Categories documented by the Federal Statistical Office Germany (2000). Whereas official statistics record waste generation, in this table the actual deposition is documented. This difference is marked by (*). Fresh weight is given unless otherwise specified. Water data refer to 1995. Unused water input refers to drainage water input to sewage treatment plants, it is therefore less than the (unknown) total amount of rainwater diverted by sealed surfaces. Population: 81.818 million.

water (for example, groundwater from mining or rain water drainage into sewage systems). The domestic input of abiotic (non-renewable) raw materials exceeds the input of biotic (renewable) inputs by a factor of 15 (fresh weight basis). ('Renewables' here refers to naturally renewable or regrowing resources; in a strict sense recyclable resources are technically renewables.)

A tremendous part of the abiotic raw material input remains unused. This is mainly due to the non-saleable extraction ('hidden flows' or 'ecological rucksacks') involved in coal mining. These masses are dumped without generating any economic utility. Landfill and mine dumping (on the output side) exceed the mass of all other waste deposited on controlled sites by a factor of more than 18. The relation of the non-used input to the input of used raw material may be used to indicate the *resource efficiency* of the corresponding extraction process (Bringezu and Schütz 1995). For example, in Western Germany from 1960 to 1990 and in Germany from 1991 to 1996, the resource efficiency of the domestic extraction of lignite decreased significantly (Bringezu 2000a).

The input of biotic raw materials from cultivation is associated with an amount of erosion that is the same order of magnitude as the raw materials. The relation of the biotic input from cultivation and the associated erosion may also be monitored over time. Bringezu and Schütz (1995) showed that the relation of harvested biomass to erosion decreased from 1980 to 1989 in Western Germany.

On the output side, CO_2 emission from fossil fuels to the atmosphere amounts to 990 million metric tons (MMT) which represents 82.0 per cent of *domestic processed output* (DPO).[2] The mass of CO_2 emission is 8.3 times higher than that of solid waste disposal. Pesticides were included because of their special importance with respect to their biocidal and metabolic disrupting potential in general. Pesticides and nitrogen and phosphorus emissions to surface waters have not been weighted by quality aspects. The amounts of release or dissipative use have been taken into account in order to lay the basis for their comparison over time.

The balance of inputs and outputs (without water) equals 0.94 billion tons. This *net addition to stock* (NAS) results mainly from the material that is added to infrastructures, buildings and so on, but also from material losses not yet considered (for example, other releases to surface waters) and it also includes statistical errors. The order of magnitude corroborates earlier studies on material flows for construction (Bringezu and Schütz 1998). Input and output are mainly determined by 'throughput flows' released to the environment after a short-term use. This applies to energy carriers, unused extraction, excavation, agricultural harvest, erosion, air and water. The extraction or harvest, as the case may be, of these materials without water comprises 81.0 per cent of domestic use of primary resources. 'Storage flows', such as construction minerals, used for long-term products and released on the output side with a certain time lag, represent a minor quantity.

In general, information can be derived from the material flow balance on the interlinkage of material inputs and outputs of the economy. Every material extracted from the environment will sooner or later burden the environment also on the output side. Any pressure related to the outputs (releases to the environment, wastes and so on) can only be diminished successfully if the input of primary materials to the economy will be reduced as well. The interlinkage of used and unused extraction, biomass harvest and erosion, waste disposal and emission from incineration may be indicated by quantitative relations which describe the metabolic profile of the economy.

A breakdown of the German material flow balance with sub-accounts for the Earth's crust (discontinuous use), cultivated soil (continuous use), air and water has been constructed (Bringezu and Schütz 1996a; Bringezu 2000a). Inputs and outputs of and between these environmental media were quantified for 1991. Transmedial flows were assessed according to volume and comparative natural flows. In central Europe in recent times there has been no natural flow comparable to the huge flow of earth crust materials in the form of fossil fuels into the air (524MMT net input). The marine dumping of 34MMT dredging materials from harbors and canals exceeded the sediment freight of German rivers to the North Sea and Baltic Sea by a factor of 24.

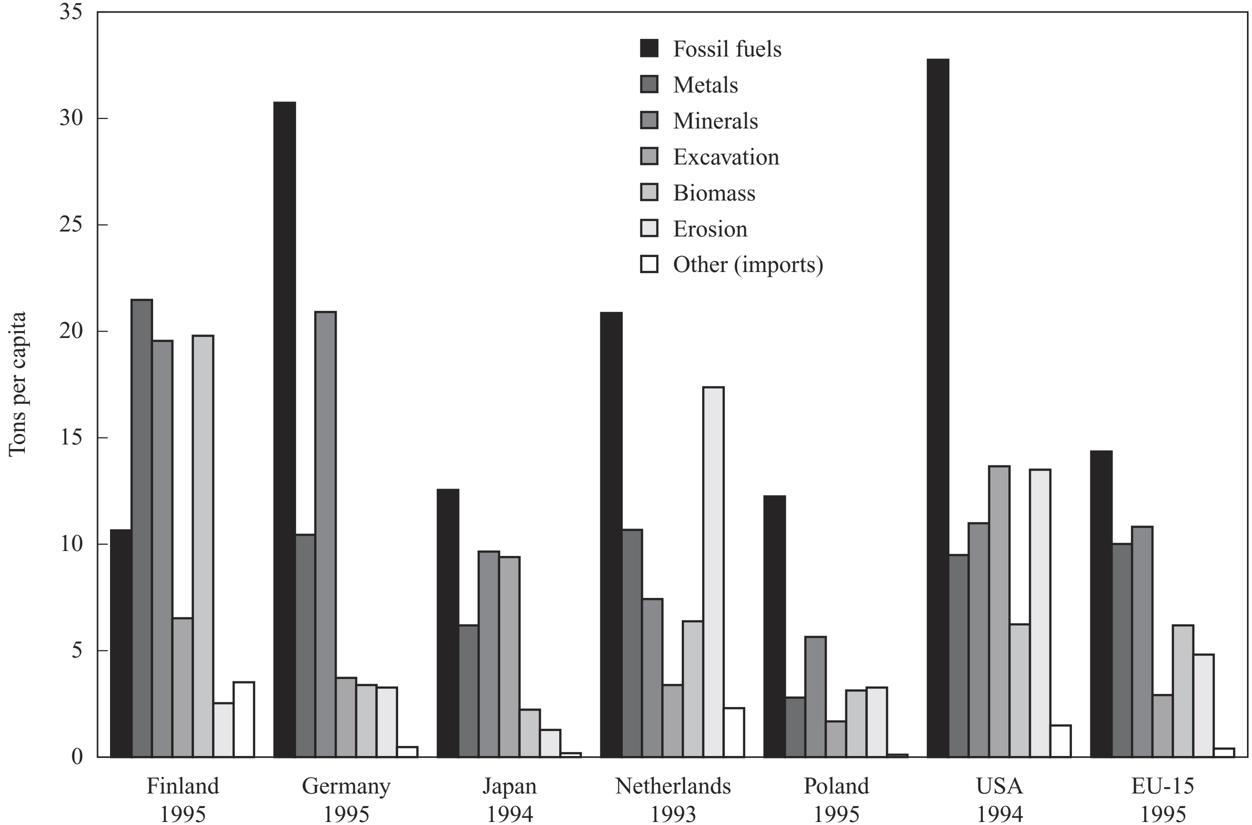
RESOURCE INPUT FLOWS

The national material flow balance does not provide information on the transnational material flows generated by an economy which burden the environment predominantly in other countries. Analysis of the upstream resource requirements of imports and exports allowed a first approximation of the *total material consumption* (TMC) of an economy (Bringezu 1993a; Bringezu *et al.* 1994; Bringezu and Schütz 1995; Bringezu *et al.* 1998b). For Germany in 1991, a TMC of the order of magnitude of 72 tons per capita (t/cap) was calculated, comprising all primary materials besides water and air. The *total material requirement* (TMR) representing the basis for national production amounted to 91t/cap. A comprehensive study on the mass and energy requirements associated with the production of aluminum, chromium, copper, nickel, manganese, phosphate and hard coal was conducted on a global mine-by-mine basis by the German Federal Agency for Geoscience and Raw Materials (Kippenberger 1999). Following the first international comparison by Adriaanse *et al.* (1997), resource flows were studied for a variety of countries (see Chapter 8).

The Composition of TMR

In 1995, the TMR for the 15 countries of the European Union (EU-15) amounted to 18.1 billion tons, or 49t/cap (Bringezu and Schütz 2001). Some 72 per cent of the EU-15 TMR was represented by resource flows of fossil fuels, metals and minerals (Figure 23.1). This averages 14.2t/cap in fossil fuel resources extracted. Energy carrier plus hidden flows amounted to 29 per cent of TMR. As a result of the lower use of energy and a reduced amount of coal use in Europe, this was only 43 per cent of the 1994 fossil fuel resource requirements of the USA. Nevertheless, in some countries such as Germany that still depend to a large extent on coal extraction, fossil fuel resource use reaches the same order of magnitude as in the USA. In the countries studied, Finland exhibited the lowest fossil fuel resource requirements.

Mineral resources are mainly used for construction. In 1995, production in EU-15 required 10.7t/cap, a level similar to the USA. Within the EU countries studied, Germany and Finland had the highest rate of mineral extraction, owing to the production of sand and gravel as well as natural stones in Germany, and the extraction of gravel in Finland. The German values were twice those of the EU-15 as a whole, owing to construction activities for houses and infrastructures which still rely on high inputs of minerals for concrete.



Note: Hidden flows are included in fossil fuels, metals and minerals or are represented by excavation and erosion.

Source: Wuppertal Institute, WRI, NIES, VROM, Thule Institute, INE and Warsaw University (see also EEA 2000; Bringezu and Schütz 2001).

Figure 23.1 Composition of TMR in the European Union, selected member states and other countries

The lowest requirements for minerals were shown for the Netherlands. Resource requirements for metals were on a higher level in the EU-15 (10.1t/cap) than in the USA (9.4t/cap). There was a significantly higher flow in Finland (21.5t/cap), where metal manufacturing still represents a relevant element of industrial production. In comparison, the metal resource requirements of Japan in 1994 were 1.6 times lower than those of the EU-15 in 1995.

At 6t/cap, biomass represented 12 per cent of TMR in the EU-15. This was only 2 per cent lower than the US biomass harvest in 1994. Most of the biomass stems from agriculture. However, Finland provided a twofold exception. First, the input of biomass amounted to 23 per cent of TMR, and second, the biomass was dominated by forestry cuts, which also represent a significant basis for the Finnish export industry. The proportion of regrowing resources in Finland was 1.9 times higher than the EU-15 as a whole.

Erosion of agricultural fields contributed only 10 per cent of the TMR in the EU-15. In the USA the amount of erosion had been reduced by policy programs, yet it was still 2.9 times the EU-15 level. Within the EU member states studied only the Netherlands were clearly above the average. This reflected the high amount of agricultural imports traded and processed in the Netherlands and associated with high levels of erosion in the countries of origin.

Foreign Resource Requirements

Domestic production of primary resources differs from imported commodities (raw materials and semi-manufactures) with regard to the related hidden flows as shown in Table 23.2. In 1995, imports of fossil fuels (excluding electricity) into the EU-15 had a significantly lower hidden flow ratio than the domestic extraction of energy resources. This resulted from the fact that the imports were mainly oil and natural gas. Those materials are associated with lower hidden flows than lignite and hard coal, which contribute significantly in some of the member states. Imports of metal resources were associated with 17 times higher hidden flows than domestic extraction. Ore mining within the EU-15 plays only a minor role. It concentrates on deposits with relatively high efficiency of extraction and lower volume burden to the environment. Most of the base metals, such as iron, aluminum and copper, are imported. Precious metals with the highest ratio of unused to used extraction are mostly brought in from outside. Between 1995 and 1997, the dominant contribution to mineral requirements came from the import of diamonds. The tiny amount of 37 to 44 tons was linked to the calculated extraction of 195 to 232MMT. This repre-

Table 23.2 Ratios of hidden flows to commodities for the EU-15 in 1995

	Domestic	Foreign	Total
Fossil fuels	3.44	1.63	2.53
Metals	0.94	16.08	11.33
Minerals	0.22	4.41	0.32
Agricultural biomass	0.62	5.90	0.88
Total	0.92	4.28	1.52

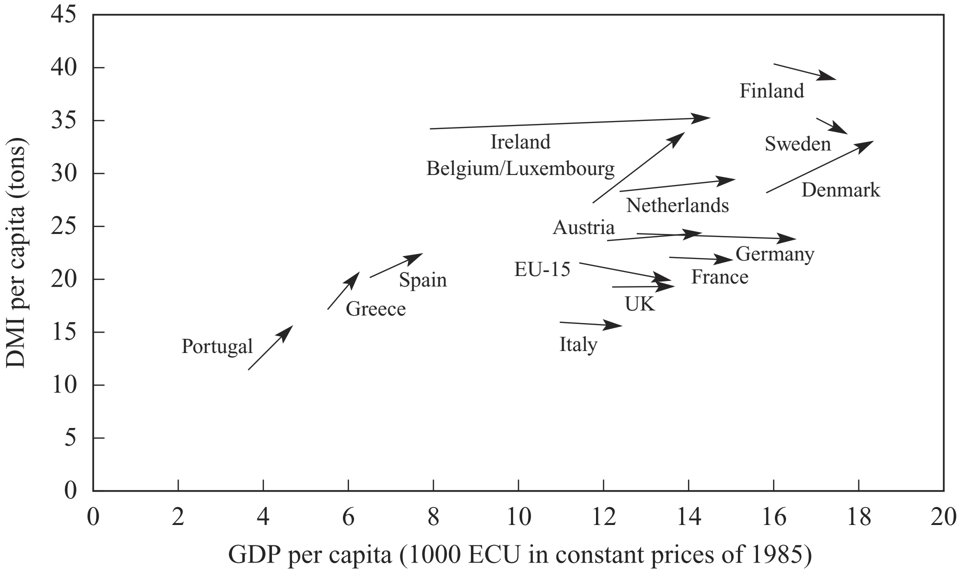
Sources: Bringezu and Schütz (2001); EEA (2000).

sented 68 per cent of the mineral resource share of imported TMR in 1997. Hidden flows due to the additional import of 2337 to 2450 tons of other precious stones have not been attributed owing to lack of data, although it is known that in some cases precious stone mining may reach the hidden flow ratio of gold ($10^5 - 10^6$).

The import of agricultural products to the EU-15 was associated with a higher amount of erosion than domestic agriculture. This resulted mainly from the import of products such as coffee and cocoa which are cultivated in tropical countries. Erosion is influenced by many parameters, such as rainfall, slope and cultivation practices. Worldwide erosion is a severe threat to soil availability and fertility and food production (Pimentel *et al*. 1995).

Direct Material Input (DMI) and Economic Development

The DMI of the EU-15 exhibited a moderate reduction in absolute terms of 5 per cent between 1988 and 1997.[3] This was equal to an 8 per cent decline from 21.2t/cap to 19.5t/cap. Most of the change occurred at the beginning of the 1990s and was mainly a result of an import decline of 1t/cap. However, from 1993 the DMI of the EU-15 followed a slightly increasing trend.



Source: Wuppertal Institute (see also EEA 2000; Bringezu and Schütz 2001).

Figure 23.2 Trend of GDP and DMI in member states of the European Union, 1988–95

Direct resource productivity (GDP/DMI) of the EU-15 increased by 28 per cent from 1988 to 1997 (Figure 23.2). Whereas in most EU countries economic growth was associated with increased DMI, reduced dependence on direct material inputs was recorded for Finland, France, Italy, Sweden and the UK. In most cases this was mainly due to reduced construction. The trend of declining DMI associated with higher levels of GDP corroborated earlier findings of Jänicke *et al.* (1992), who studied the consumption of selected materials in industrialized countries.

OUTPUT FLOWS TO THE ENVIRONMENT

Time series of socioeconomic and material flow parameters in Germany reflect the shift from West Germany in 1990 to the reunited Germany in 1991. For comparison, data are shown on a per capita basis since in Germany as a whole after reunification the population was 26 per cent and GDP 24 per cent higher than in Western Germany before. (In the FRG, the population had been rather constant over the whole period from 1975 to 1990. After 1991, the German population increased by 2.6 per cent until 1996.) From 1975 to 1996, DPO was almost constantly high in Western Germany as well as in the reunited Germany, with values around 15t/cap. In contrast, total domestic output was significantly increased in reunited Germany, to 54t/cap, compared to about 30–40t/cap in West Germany. This was due to the lignite[4] mines in the eastern part, which had been the backbone of the energy supply in the former GDR.

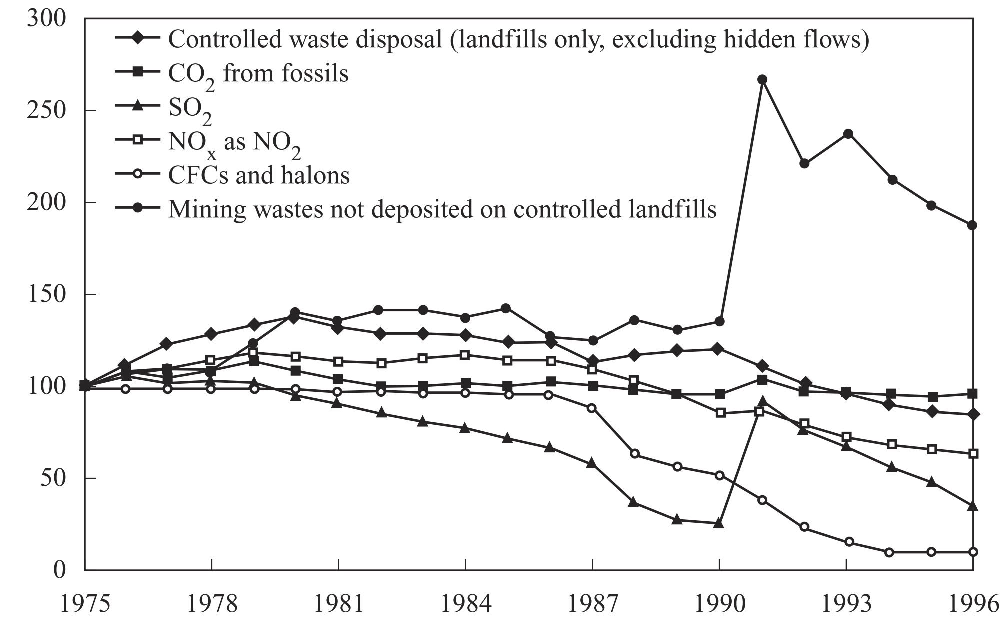


Figure 23.3 Temporal trends of selected per capita material output flows (Index 1975 = 100) in Germany (West Germany 1975–90, reunited Germany 1991–6)

Interesting temporal trends were recorded for some selected material flows (Figure 23.3). Mining wastes for landfills (not disposed of in controlled sites) increased with reunification and decreased afterwards until 1996. This was due to a phase-out of several lignite mining facilities in the eastern part of Germany. However, lignite mining is still going on in Germany. (It has been the center of regional political debate because single villages had to be abandoned in favor of mining excavation.) Most of the associated hidden flows are recorded in official German statistics.

Among the declining trends of material outputs, that of CFCs and halons is especially obvious. As with SO_2 emissions, these are examples of declining material outputs due to effective policy regulations. Nevertheless, the constant DPO level indicates that the overall output flows have not been reduced by regulation. Between 1975 and 1996, CO_2 emissions from fossil fuels ranged from 84 per cent to 87 per cent of DPO. This represents the most dominant volume of processed outputs of the economy to the environment. In 1996, 89 per cent of DPO was released to the atmosphere, the 'globalized waste bin'.

PHYSICAL GROWTH OF THE ECONOMY

Physical growth of the economy is indicated when the volume of input flows of the technosphere exceeds the volume of the output flows. For Germany at the beginning of the 1990s, net addition to stock (NAS) was calculated as about 10t/cap (Bringezu and Schütz 1995). NAS relates to additional buildings and infrastructures. It was determined by balancing the inputs and outputs of the national MFA (Table 23.1) and cross-checked by direct accounting of the material flows for construction. NAS may be regarded as an indicator of the distance towards a flow equilibrium between input and output of an economy. Flow equilibrium is regarded as a necessary condition of a sustainable situation (Bringezu and Schütz 1998; see also Chapter 8). First international comparisons of NAS were conducted by Matthews *et al*. (2000). Between 1975 and 1996, the order of magnitude remained close to constant within the studied countries, although there was a variation between countries (Table 23.3).

Table 23.3 Net addition to stock indicating the physical growth rate of the economy

	1975	1995
Austria	9.69	11.42
Germany	*12.20	11.84
Netherlands	11.25	8.75
Japan	8.24	9.48
USA	7.18	7.43

Note: *Western Germany.

Source: Matthews *et al.* (2000).

SECTORAL ANALYSES

A first comprehensive physical input–output table (PIOT) was established for Western Germany in 1990 (Radermacher and Stahmer 1998; Stahmer *et al*. 1997) and is going to be provided every five years by the German Federal Statistics Office (FSO 2000). The German PIOT comprises product flows between sectors and resource inputs from the environment (used and unused extraction) plus emissions and waste disposal. On the input side the substance input for biomass production (such as carbon dioxide) is accounted for and corresponds to emissions of the same substances on the output side. Until now, economy-wide material flow balances have accounted for the harvested biomass (for practical reasons). National material flow balances defined the content of waste deposits as being outside the anthroposphere. At present, the PIOT assigns the input of waste deposits to man-made assets, a circumstance which may lead to equivocal interpretations of NAS.

The attribution of indirect resource inputs to the sectors of intermediate and final demand was performed on the basis of economic input–output (I/O) tables (Bringezu *et al*. 1998b). Final demand as defined by I/O statistics comprises private consumption, state consumption, investments, exports and storage changes. In 1990 (for the Federal Republic of Germany), the construction industry, manufacturing of metals, the construction of vehicles, vessels and aeroplanes and the energy sector provided most material-intensive products when considering direct and indirect resource requirements. Based on current technology, the relative dependence on a material-intensive supply was greatest in the energy supply sector, the iron and steel industry and the construction sector. The attribution to main fields of private demand showed that housing, nutrition and leisure were most resource-intensive. Between 1980 and 1990, TMR of the FRG was rather constant, while GDP increased significantly. Decomposition analysis revealed that the increase in total resource productivity (GDP/TMR) was mainly due to changes in technology (Moll *et al*. 1999).

The outputs of industrial sectors to the environment were analyzed for various substance emissions and waste and waste water categories. A comprehensive German emission inventory was established (FSO 2000) which can also be used to model indirect emissions to certain sectors (Hohmeyer *et al*. 1997). At the European level a similar but less detailed inventory was established. Those data ('ETC air quality and emissions') can be accessed at *http://www.eea.eu.int*.

Scenarios for sustaining the European economy have been analyzed using the energy-based ECCO model which also includes primary material requirements (Spangenberg and Scharnagl 1998). For Germany, econometric modeling comprising physical inputs and outputs from and to the environment, was applied for a study on labor and ecology (HBS 2000). This model is based on the Pantharei model (Meyer and Ewerhart 1998a).

Construction flows of the German economy have been quantified (Bringezu and Schütz 1998; Bringezu 2000a) and analyzed with regard to future scenarios (Kohler *et al*. 1999). A dynamic model for construction flows has been developed to simulate the demand for resources depending on different types of construction (Buchert *et al*. 1999). The Pantharei model was used to assess scenarios of renovation through programs enhancing energy efficiency (for example, through insulation of existing buildings) with regard to natural resource requirements, climate gas emissions, necessary investments and implications for employment (Wallbaum 2000).

MATERIALS AND PROCESS CHAIN-ORIENTED ANALYSES

The flow of selected materials and substances through the German industry and beyond has been studied by a variety of researchers. For instance, there is a study of the flow of aluminum through production and consumption and interlinked material flows with reference to the development of integrated environmental and economic statistics (Bringezu *et al.* 1998a) and a special research program on resource-oriented analysis of metallic raw material flows (Kuckshinrichs *et al.* 2000).

Nutrient flows such as nitrogen, phosphorus and potassium have been balanced to assess agricultural performance (Bach and Frede 1998). Extended modeling was used to predict environmental loads (Behrendt *et al.* 1999). Various studies have been integrated into an overall flow assessment of nitrogen (ATV/DVWK 2000) in order to support priority-oriented political action for which targets had already been formulated (BMU 1998).

The flow of hazardous chemicals such as cadmium and chlorinated substances as well as flows for the production of textiles and cars were studied on behalf of the Enquête Commission (1994, 1998). The flow of PVC was taken as an example to discuss possible criteria for the assessment and the main fields for materials management (UBA 1999).

For the assistance of 'bottom-up' analyses, physical inputs and outputs of certain (unit) processes have been provided in computer-based models such as GaBi (*http://www.gabi-software.com*), GEMIS (*http://www.oeko-institut.de/service/gemis/index.htm*) and UMBERTO (*http://www.umberto.de*), originally designed to simulate the emissions for certain process chains, but which also include some categories of resource requirements and may be used for LCA, firm-related accounts and MFA as well.

STUDIES ON REGIONS AND INDUSTRIAL NETWORKS

The metabolism of the old industrialized Ruhr region has been analyzed using comprehensive material flow balances, sectoral attribution and disaggregation down to the community level (Bringezu and Schütz 1995; Bringezu 1999; Bringezu 2000a). Within the different communities, mining and manufacturing underwent technological change which tended towards increased resource efficiency, but in significantly varying degrees.

Specific issues of the metabolism of regions have been studied as tools for enhanced regional materials management (Thrän and Soyez 2000; Thrän and Schneider 1998). The management of regrowing resources from forestry and agriculture and the build-up of regional producer–consumer networks have also been assessed from the point of view of assessing regional production of value added. For example, the potentials and options of increased use of timber products were studied in the Trier region (Maxson *et al.* 2000) and in the Ostprignitz–Ruppin region of the state of Brandenburg (Thrän and Schneider 1998). The establishment and extension of the use of fiber products from agriculture is being studied for the Dresden region (*http://www.nachhaltig.org/ghkassel/prolang.htm*).

Studies on industrial networks have been performed for residues of metal manufacturing in the Ruhr (Schwarz *et al.* 1996). Management issues for cooperation chains along the production–consumption route were discussed from various perspectives in Strebel and Schwarz (1998). In 1998, the German Federal Ministry for Education and Research launched a research program on 'model projects for sustainable economies'. The main topics include 'agriculture and regional marketing', 'regional material flow management' and 'strengthening regional potentials' (see *http://www.nachhaltig.org*). First results of the regional flow management studies are available (Liesegang *et al.* 2000). Optimization of waste use within Heidelberg's Pfaffengrund industrial area and the surrounding Rhine–Neckar region was studied to improve communication structure and to provide on-line tools for materials management in and between companies (Sterr 2000). Work is going on at present on metal manufacturing in the Hamburg region (Gottschick and Jepsen 2000). At the industrial park level, cooperation between companies for utilization of waste materials and energy was studied in Henstedt–Kaltenkirchen northeast of Hamburg (Grossmann *et al.* 1999). This work was supported by the environmental ministry of the state of Schleswig–Holstein which issued a guide for cooperation and networking (MENFSH 1999). A data base of appropriate information for a materials and energy network is being built for the Karlsruhe Rhine harbor (Fichtner *et al.* 2000). For a comparison of industrial network studies in the German-speaking area, see Wietschel and Rentz (2000).

In 1997, the German Federal Office for Building and Regional Planning (FOBRP 1999) initiated a competition between 26 'regions of the future'. The results were presented at the Urban 21 conference in Berlin in 2000. Several pilot projects designed to create more efficient materials and energy flows were also conducted within those regions (*http://www.bbr.bund.de/english/moro/future.htm*).

At the community level different approaches have been applied in various cities for environmental reporting and management. In Germany, the 'Eco-Budget' method was developed and tested by communities like Dresden, Heidelberg, Bielefeld and the region of Nordhausen (Burzacchini and Erdmenger 2000 and *http://www.iclei.org/ecobudget*). Analogously to financial budgeting, the community council decides on an 'environmental budget' on the basis of physical information on the consumption of resources and the release of emissions and generation of waste, both in absolute terms and in relation to the economic (and social) performance.

NOTES

1. Institutes of the German-speaking region present their projects in the Working Group on Material and Energy Flow Accounting, *http://www.statistik-bund.de/mv/agme.htm*.
2. DPO comprises all outputs besides water and hidden flows such as landfill and mine dumping and erosion.
3. For this time series all the countries in the EU-15 since 1995 were included. The European Community of the EU-12 grew in 1990 through reunification of Germany and again in 1995 when Austria, Finland and Sweden joined.
4. Lignite is one of the most resource-intensive energy carriers. In 1996, 9.34 tons overburden and non-saleable production were extracted to produce 1 ton of lignite.

24. Material flow analysis and industrial ecology studies in Japan

Yuichi Moriguchi

Although the term 'industrial ecology' (IE) itself has not been very widely used in Japan, numerous studies and practical efforts have been undertaken in related fields, and those activities have been expanding rapidly. The term 'IE' itself, and its concept, tools and applications are also being disseminated. A textbook of IE has been translated into Japanese and published (Gotoh 1996). Japanese activities in this field have been disseminated through international journals (Moriguchi 2000) and other publications. Industries and universities as well as national research institutes have been playing active roles. This chapter will briefly review IE studies in Japan, then characterize Japanese material flows based on international joint studies.

BACKGROUND

Japan experienced severe environmental pollution involving serious health damage from the 1960s to the 1970s, the era of rapid industrialization. End-of-pipe technologies for large point sources, such as desulfurization and denitrification, have successfully contributed to diminishing such traditional environmental pollution problems. Can we continue to rely on such end-of-pipe approaches to solve all of the emerging environmental problems at the end of the huge energy and material flows of the industrialized economy?

The answer seems to be rather negative. Many of the present environmental issues have their roots in the basic structure of industrialized society, characterized by mass-production, mass-consumption and mass-disposal. There is a need to transform production and consumption behavior to more sustainable patterns. Recognition of this is clearly stated in recent Japanese national environmental policy documents such as the Basic Environment Plan. Based on this, a basic law for establishing the recycling-based society was enacted in 2000 concerning more sustainable material management. It seems that looking upstream is at least being built into environmental policies.

Such a paradigm shift in Japanese environmental policy may be interpreted differently in global and domestic contexts. Increasing attention is being paid to global environmental problems, and the concept of sustainable development is being spread. Recognition that the environment is finite as a source of resources supply and as a recipient of residuals is the most essential standpoint to discuss sustainable development. At the local level, on the other hand, limitation of the end-of-pipe approach is becoming evident (Moriguchi 1999). It has to be kept in mind that such recognition in Japan cannot occur unless the public is aware of urgent visible problems with municipal solid wastes (MSW) and industrial wastes. Japan is suffering from the shortage of final disposal site capacity, but the development of new dumping sites is difficult because of potential negative impacts on the environment. The cost of dumping industrial solid wastes is high enough to call industry's attention to waste minimization. Incineration of solid waste has been effective in decreasing final disposal, but the recently revealed problem of dioxins from waste incineration is another force bringing people's attention back to the negative aspects of the mass-disposal society.

OVERVIEW OF MFA AND OTHER IE STUDIES

Material Flow Analysis/Accounting

As will be described later in more detail, Japan depends highly on imported natural resources, which often have environmental relevance. Harvesting of timbers from tropical rainforests is a typical issue. Japanese participation in the OECD pilot study on natural resource accounting (NRA) at the beginning of the 1990s (OECD 1994c) was driven by concerns over this. The experience of the forest resource account was later applied to Asian countries (Koike 1999).

On the other hand, material flow analysis (MFA) was studied mainly in order to respond to the domestic issues of increasing solid wastes. A flow chart describing Japan's macroscopic material flow balance has been published in the annual *Quality of the Environment* report since 1992 (EAJ annual). Dissemination of an English edition of the report created the opportunities for European experts on MFA to involve Japan in international collaborative efforts in this field. In 1995, the SCOPE (Scientific Committee for Problems on Environment) organized a scientific workshop for indicators of sustainable development at the Wuppertal Institute in Germany. Participants from four industrialized nations, Germany, the USA, the Netherlands and Japan, agreed to launch an international collaborative study to compare their overall material flows at the national level. The results will be shown later in this chapter.

Interindustrial flows of some individual materials such as non-ferrous metals have been studied mainly from the viewpoint of material recycling (Clean Japan Center 1997). Substance flow analysis (SFA), which captures the flow of specific elements of environmental concern, was applied to some case studies, such as with an analysis of nitrogen flow and its impacts on eutrophication. The SFA framework for toxic substances has yet to be explicitly adopted.

Inventories of pollutants emissions (that is, 'emission inventories') may be categorized as one specific form of MFA in a broader sense. Official inventories are compiled for greenhouse gases (GHGs) in accord with an international convention, whereas those for others, even for traditional air pollutants, have not been made available by authorities until recently. This is mainly because the institutional basis for environmental statistics is rather weak. A PRTR (Pollutant Release and Transfer Register) system was tested in pilot studies in the late 1990s, and a nationwide system under the newly enacted law started in the year 2001.

Life Cycle Analysis/Assessment

Life cycle analysis (LCA) has been studied since the 1970s in Japan, although the term 'life cycle assessment' has been applied recently with a far stricter definition. The main concern was the analysis of life cycle energy consumption (called 'energy analysis' by Kaya 1980). Life cycle energy (LCE) was studied not only in energy systems such as electric utilities using different fuel sources, but also with respect to products such as clothing, food and housing, in order to understand the overall structure of energy use from the viewpoint of final demand for commodities and services.

Although such life cycle studies were not actively pursued in the 1980s, they attracted renewed interest in the early 1990s, when concern about climate change was increasing. $LCCO_2$ (life cycle CO_2 emission) studies, which derive from LCE methodologies, were applied to various subjects, for example motor cars (Moriguchi *et al.* 1993), transport systems and infrastructure. In parallel, LCA developed a more rigorous framework (see Chapter 12) through the activities of international organizations such as the Society for Environmental Toxicology and Chemistry (SETAC) and the International Standards Organization (ISO). Newer LCAs are, as a result, typically more detailed, and product-oriented. These two streams still exist within the Japanese LCA community. An important component of Japanese LCA activity is the International EcoBalance Conference, held four times (every two years) since 1994 in Tsukuba. It has been providing exceptional opportunities for information exchange, not only internationally, but also across sectors and disciplines domestically.

The LCA national project was launched in 1998 under the leadership of the Japan Environment Management Association for Industry (JEMAI) with financing from the former Ministry of International Trade and Industry (MITI) (Yano *et al.* 2000). More than 20 industrial associations have participated in the inventory sub-project and other subprojects such as those on databases and impact assessment have been undertaken in parallel.

Input–Output Analysis with Environmental Extension

Application of input–output analysis (IOA) to environmental concerns was undertaken by Wassily Leontief, the pioneer of IOA, in the early 1970s (Leontief [1973] 1986). He carried out case studies for Japan (Leontief [1972] 1986) and this might be one reason why environmental application of IOA has been and continues to be active in Japan. The Japanese IO table consists of some 400 sectors and is thought to be one of the most detailed and qualified in the world.

IOA was already applied for the above-mentioned energy analysis, in the late 1970s. Energy consumption by sectors is indicated in physical unit tables, which officially accompany the national IO tables. Other official statistics on energy consumption are often used in order to supplement the data in the physical unit tables, in which data coverage and accuracy are not complete. Once sectoral direct energy consumption per unit of output is quantified, one can easily calculate overall sectoral energy intensity, including indirect energy consumption in upstream industries, by applying the Leontief inverse matrix. This calculation process has been applied to energy consumption, CO_2 emission (Kondo *et al.* 1996; Kondo and Moriguchi 1997), traditional air pollutants (SOx, NOx) (Hondo *et al.*

1998), water pollutants (BOD, N, P), solid wastes and so on. They have often been used for life cycle inventory analysis. Another example is an analysis of structural changes of CO_2 emissions from the viewpoint of final demand of the economy as influenced by international trade (Kondo *et al.* 1998).

More recently, application of IOA to the issues of waste management and recycling has become very active. A Waste Input–Output (WIO) model was proposed (Nakamura 2000) to describe an interdependence of goods-producing sectors and waste management sectors, in which both monetary and physical flows were dealt with. Description of whole material flows within the economy and their interaction with the environment has also been attempted, linking sectoral IO studies and macroscopic MFA studies. Learning from the German pioneering experiences in PIOT (physical input–output tables: see Stahmer *et al.* 1998), a framework of 3DPIOT (three-dimensional PIOT) is being proposed (Figure 24.1), and case studies are being undertaken (Moriguchi 1997). These environmentally extended IO studies have many features in common with MFA studies.

Environmental Indicators and Accounting

Environmental indicators and environmental accounting are the basic relevant tools for IE studies. Although there are a great many past studies on environmental indicators for the application to environmental policies in Japan, this topic is not reviewed here. Environmental indicators attract more up-to-date and urgent concerns from another user, namely, industries. Environmental performance indicators (EPI) are actively studied as a part of environment management tools.

Monetary environmental accounting at the microeconomic level, that is, corporate environmental accounting, has become widely used since the former Environment Agency of Japan published guidelines in 1999. Many leading companies disclosed their trial of environmental accounts around this period. In many cases, physical accounts were not included, but some companies published overviews of physical material flows around their activities as a part of environmental reporting.

Environmental accounting for the macroeconomy has been studied during international discussions of the Integrated System of Environmental and Economic Accounting (SEEA). Worldwide experts in this field gathered for a meeting in Tokyo in 1996 (Uno and Bartelmus 1998), which stimulated the international exchange of experiences. The former Economic Planning Agency of Japan published its preliminary calculation of environmentally adjusted net domestic product (EDP). Though this monetary accounting study and the physical material flow accounting study led by the author have been undertaken within the same research project, they have not yet been fully integrated.

Other IE Studies and Initiatives

Japan leads the world in the share of companies receiving ISO 14001 certification. Other tools of the ISO 14000 family have also been studied and/or implemented. In addition, several research initiatives in Japan have been and are currently under way in designing environmentally sound products and materials. Projects variously called ecodesign, design for environment (DFE) or ecoproducts belong in this group. Also what is often called ‘inverse manufacturing’ has been proposed and studied to promote the design for

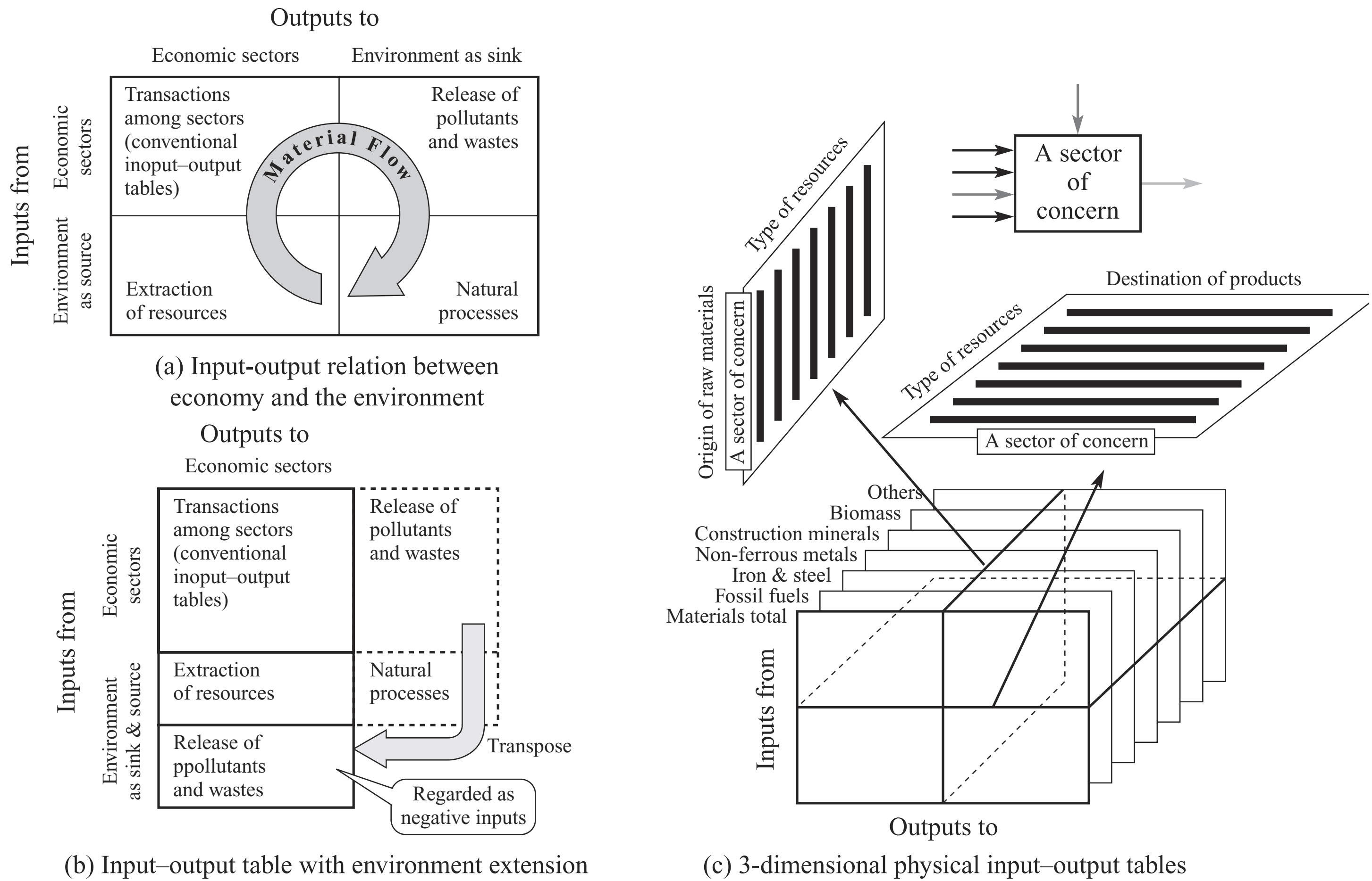


Figure 24.1 Frameworks of environmentally extended physical input–output tables

disassembly (DFD), and this is converging with ecodesign initiatives. The Union of EcoDesigners was established with participation of a number of related academic communities in Japan. Many visitors attended an exhibition of ecoproducts held in Tokyo. On the basis of these successful endeavors, partnership between academic efforts in ecodesign and business opportunities arising from ecoproducts appears to have been strengthened.

Ecomaterial research, originally launched by the former National Research Institute for Metal in the early 1990s, is another active field of inquiry. This research aims at improvements in the eco-efficiency of materials. A crucial focus of this work is the improvement in functionality of materials along with a reduction in the environmental impacts of the materials in order to increase eco-efficiency.

Issues of extended producer's responsibility (EPR) are discussed mainly in the context of post-consumer waste management. As stated earlier, promotion of recycling is often encouraged as a means of waste minimization. Based on EPR thinking, new national legislation for recycling of household electric equipment (air conditioners, TVs, refrigerators and washing machines) was put in force in the year 2001.

Zero-emission initiatives, which focus on re-use of wastes of one industry by another to achieve waste minimization, are not only studied by researchers but also practiced, at least on a trial basis, in various sectors, such as machinery and the food and beverage industries. Many university researchers have participated in zero-emission studies since 1997 via funding from the Ministry of Culture and Education. Other research projects are also supported by governmental funding. For example, the Construction of recycle-oriented industrial Complex systems with environmentally sound technology at societal experimental sites Project (CCP) funded by the Japanese Science and Technology Agency is a new attempt at an empirical field study in industrial ecology. The CCP project focuses on three topics: industrial parks, urban renewal and the food system.

A concept closely related to the IE, 'industrial transformation' (IT), developed under the auspices of the International Human Dimension Program (IHDP), has also attracted interest in Japan and provided an opportunity to exchange experiences regarding industrial ecology and socioeconomic research.

CHARACTERIZATION OF JAPANESE MATERIAL FLOWS

Geographical and Historical Background

Japan is very densely populated, with a population of about 125 million, whereas the domestic stock of natural resources is not sufficient and its exploitation to sustain this population is costly. Therefore Japanese material flows should not be discussed without considering international trade flows. The history of international trade in Japan has an interesting profile. The Edo era, when Japan was closed to foreigners and international trade, is often referred to as a model environmentally sustainable society because of its self-sufficiency in resources. In contrast, the present Japanese economy heavily depends on international trade, both imports and exports. Without a tremendous amount of imported natural resources, such as fossil fuels and metal ores, the Japanese economy cannot be sustained. The growing export of products by raw material industries and assembly industries has been a major driving force of rapid economic growth.

In the following sections, Japanese material flows will be characterized, on the basis of a database recently compiled through participation in the international joint project on MFA (Adriaanse *et al.*, 1997, Matthews *et al.* 2000; see also Chapter 8). In this joint study, in addition to direct material inflows and outflows, hidden flows (originally named 'ecological rucksacks' in Germany) were quantified. They refer to ancillary material and excavated and/or disturbed material flow, along with the desired material.

Material Inflows

An overview of Japanese material flows is shown in Figure 24.2. Material input flows that support the Japanese economy are characterized by high dependency on the import of natural resources. Imported commodities account for about one-third of the mass of direct material inputs (DMI) to the economy, and account for one-half of the total material requirement (TMR), which is the sum of the DMI and hidden flows.

Imports provide the Japanese economy with essential materials, including fossil fuels, metal ores, and agricultural and forestry products. Import dependency is particularly high for metal ores and fossil fuels, as there are only poor stocks of these resource categories from domestic sources. Dependency is also high for timber, in spite of the fact that two-thirds of Japan is covered by forest. Timber imports constitute a significant portion of the entire international trade of the world. Recent trends have revealed that commodities increasingly tend to be imported in more manufactured form, for example, refined metals rather than metal ores, or plywood rather than roundwood. Imports of semi-manufactures and final products have also been increasing. Large hidden flows are associated with metals (particularly with copper and iron) and coal, as well as agricultural and forestry products.

Domestic material flows, both commodity mass and 'rucksacks', are dominated by construction activities, nearly 90 per cent in early 1990s. Domestic construction minerals including limestone, crushed stone and sand and gravel are used to create and improve buildings, roadways, water reservoirs and other infrastructure.

Material Outflows

The largest output flow from the economy to nature is the emission of carbon dioxide. Apart from the fact that CO_2 is notorious as a greenhouse gas (certainly a matter for concern) we have to recognize another fact: CO_2 is the heaviest waste from the industrialized economies.

After CO_2, waste disposal to controlled landfill sites is the next major component of direct processed output (DPO). This is of greater environmental significance than the nominal weight implies. Japan has a shortage of landfill sites for waste disposal. Reclaiming coastal areas for this purpose has sometimes caused the decrease of habitats for wildlife. The amount of waste disposal to landfill is much smaller than the amount of waste generated. The difference between the amount generated and the amount sent to landfill is the amount recycled or reduced by incineration and drying. Three-quarters of MSW is incinerated to reduce waste volumes. The amount of landfill wastes was almost constant until 1990, but is now decreasing, thanks to waste minimization and recycling efforts.

Input of food and feed is balanced by return flows of CO_2 and water after digestion.

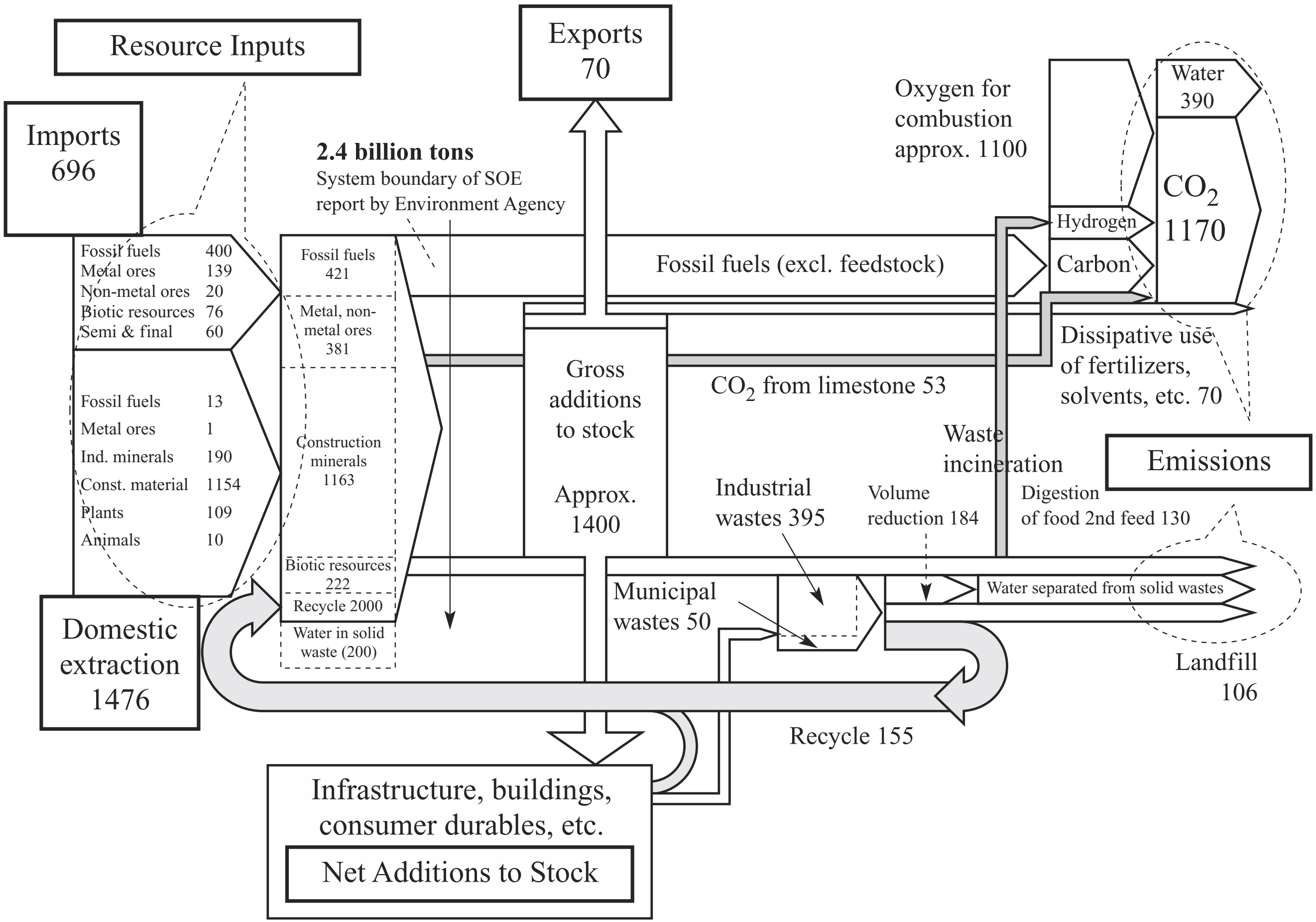


Figure 24.2 Materials balance for Japan, 1990 (million metric tons)

Dissipative use is another important category of output flows. Dissipative flows are dominated by applications of animal manure to fields. Fertilizers and pesticides are intensively used in Japanese agriculture to enhance productivity and compensate for the limited area of available farmland.

However, the total of these output flows is still much less than input flows. This is because more than half of direct material inputs is added to the stock, including consumer durables, capitals of industries and public infrastructures. They can be deemed potential sources of waste in the future.

Domestic Hidden Flows

Soil excavated during construction activities dominates domestic hidden flows. Only surplus soil, which means the soil excavated and then moved out of the construction site to landfill or other sites for application, is quantified by official surveys. The total quantity of soil excavation by construction activities is much greater, because excavation work is usually designed to balance 'cut and fill', to use excavated soil on site and minimize the generation of surplus soil.

Hidden flows associated with mining activities are trivial in quantity in Japan, because of the limited resources of fossil fuels and metal ores. Consequently, the contribution of domestic hidden flows to total domestic output (TDO) is relatively small, compared with more resource-rich countries. It should be borne in mind that the small size of domestic hidden flows is counterbalanced by imported hidden flows associated with imported metals and energy carriers; this represents the transfer of Japan's environmental burden to its trade partners, which the first joint report emphasized (Adriaanse *et al.* 1997).

International Comparison of MFA-based Indicators

Japanese TMR is about 45 tons per capita, which is much lower than the other three countries studied (around 85 tons per capita). This is mainly because of smaller energy consumption per capita and lower dependency on coal. In terms of DMI per capita, the Japanese figure is only slightly smaller than those of Germany and the USA. DMI per capita for the Netherlands is also on a similar level, if huge transit import flows to other European countries are excluded.

Dependency on imported material flows varies largely among countries, from less than 10 per cent for the USA to 70 per cent for the Netherlands. Ecological 'rucksacks' accompanied by imports imply various environmental impacts on trade partners. More specific analysis will be necessary to identify individual problems behind ecological rucksacks. The absolute level of DPO per capita in Japan is about 4 metric tons without oxygen and 11 metric tons with oxygen. These values are relatively small in comparison to the countries studied.

Recent Trends

According to the analysis of historical trends of indicators, Japanese TMR per capita is trending upwards. This coincides with the increase of waste flows. DPO and TDO in Japan grew 20 per cent during the period 1975–96. This growth occurred mainly after the late 1980s. Before then, DPO was almost constant and TDO decreased slightly. On a per capita basis, there was a downward trend in TDO from the late 1970s to the mid-1980s. DPO per capita also decreased slightly in this period. Growth in DPO per capita and TDO per capita were particularly evident in the late 1980s, when the country experienced the so-called ‘bubble economy’.

Material output intensity, that is, DPO or TDO per constant unit of GDP, declined until 1990 because of larger growth in the monetary economy than in physical throughput (the physical economy). However, since 1990 DPO and TDO have continued to increase while economic growth has slowed down. This recent trend can be explained by structural changes in energy consumption in part due to relatively low oil prices. Household energy consumption including fuel consumption by private cars has increased and contributed to larger CO_2 emissions, but this trend has contributed little to GDP growth.

Net additions of materials to the stock (NAS) in the Japanese technosphere have fluctuated in accordance with patterns of governmental and private investment. NAS increased significantly in the late 1980s, then stabilized at a lower level. Because Japan has a shorter history of industrialization than Western countries, construction work is still active and contributes significantly to the country’s overall picture of material flows. As much as 60 per cent of direct material input (DMI) is added to the stock. This figure also has a close relation to inputs of construction materials as well as to soil excavation. Increasing quantities of stock imply that demolition wastes will also increase in the future. Currently, the government is attempting to encourage recycling of demolition wastes.

CONCLUSION

Many different industrial ecology tools have been studied and applied in Japan, though they have not necessarily been reported internationally. Linkage among different tools within industrial ecology, integration of engineering tools and economic analysis and linkage between tools for microeconomic analysis and those for macroeconomic analysis should be further elaborated. Physical accounting for material inflows and outflows at any level can serve as a common basis. International exchange of experiences, as in the case of the MFA study, will contribute to further developments and improvements in industrial ecology methodologies and practices.

25. Industrial ecology: an Australian case study

Andria Durney

This chapter presents an application of the industrial ecology concept at the national level, using Australia as a case study. Australia is an industrial country that is also one of the world's biggest natural resource exporters. This circumstance provides a sharp contrast with, for instance, the UK or Germany.

Before beginning, some clarification is needed on the usage in this chapter of the concepts of industrial ecology and industrial metabolism. The industrial metabolism framework can be used to identify the sources and sinks of major material and energy flows resulting both directly and indirectly from economic activities, and to estimate the magnitude, rate, composition and direction of these flows (see, for example, Wolman 1965; Lutz 1969; Stigliani *et al.* 1994). This information can then be used to assess the environmental impact of these materials/energy flows and the possible political, economic, technological, social and other forces driving them. Industrial ecology encompasses a broader range of issues than industrial metabolism since it can potentially consider *all* economies – both 'developed' and 'developing' – as well as a wider range of anthropological forces inducing industrial material flows (Socolow *et al.* 1994; Allenby 1992b). Industrial metabolism methodologies are therefore valuable to use within the broader industrial ecology concept. Both industrial metabolism and industrial ecology approaches can also consider the ecological importance of unpriced material flows, such as overburden from mining (Ayres and Kneese 1969; Schmidt-Bleek and Bringezu 1994). See Chapters 1 and 2 for more details of the way the industrial metabolism and industrial ecology approaches relate to major environmental theories.

PURPOSE OF AN AUSTRALIAN INDUSTRIAL ECOLOGY CASE STUDY

There are three main reasons for providing a case study of Australia's industrial ecology: (a) to demonstrate the methodology of the industrial ecology concept at the national level; (b) to indicate possible major factors influencing a nation's industrial ecology; and (c) to illustrate data and research needed to evaluate and improve a nation's industrial ecology.

Australia is chosen for its unique physical, cultural and structural aspects, to emphasize the corresponding need for a unique focus of industrial ecology methodology and instruments for each country. Information from national case studies is also a vital step towards the long-term goal of sustainability for the whole planet. Meadows *et al.* (1972) and WCED (1987) eloquently argue the need for global ecological sustainability.

METHODOLOGY

The methodology of this Australian case study is based on work done by major authors in the field of material accounting and ecobalancing (for example, Ayres and Kneese 1969, 1989; Ayres and Rod 1986; Ayres 1989b; Baccini and Brunner 1991; Steurer 1992; Stigliani *et al.* 1994; Schmidt-Bleek 1993b; Ayres and Simonis 1994). Ideally, at least the following five major steps should be undertaken to estimate Australia's industrial ecology.

1. Identify and quantify major material and energy inputs from the environment to industry, and outputs from industry to the environment within Australia's borders. Both diffuse and point emission sources (Stigliani *et al.* 1994) as well as indirect flows should be considered.
2. Trace the path of material flows within industries and between industries and the environment using mass balance principles on a cradle-to-grave basis, within Australia's boundaries.
3. Calculate material stocks for both products and infrastructure (Baccini and Brunner 1991).
4. Find Australia's material intensity by estimating its total material consumption and total material emissions (Ayres and Kneese 1968b, 1969, 1989; Ayres 1989b; Billen *et al*. 1983). The level of recycling, waste mining, dematerialization and dissipative use can all help assess the sustainability of Australia's current industrial ecology.
5. Identify key anthropological forces driving significant material flows to facilitate planning towards improved industrial ecology in Australia.

Unfortunately, Australian data are not available in as much detail as in the USA and some European countries. Major international sources of environmental and industrial data have been used in this chapter, but many of the Australian data were very old – early 1970s – or non-existent. Data are also lacking worldwide on outputs of industrial processes, especially waste flows, and some inputs (Stigliani and Jaffe, 1993; Schmidt-Bleek 1993a). (For a discussion of the use of process analysis to fill in some of these gaps, see Ayres 1978.) Hence this chapter presents simply a preliminary rough sketch of Australia's industrial ecology, which can be extended and improved as more data become available.

RESULTS AND DISCUSSION

In this section major inputs, outputs, paths of material flows and materials stocked in Australia's industrial system are considered quantitatively wherever possible. A rough indication of Australia's national material intensity is given and possible key forces shaping Australia's industrial ecology are identified.

Australian Industrial Ecology Inputs

Agricultural, forestry and mineral material inputs into Australian industrial activities are considered to be direct inputs from the environment into the industrial system (except for fertilizers and pesticides) and thus represent the cradle of industrial material flows.

Agricultural inputs

Agricultural inputs are defined here as food plants and animals, fertilizers and pesticides, and land and water used for agriculture. In particular, Australia has a very high level of meat (including seafood and poultry) and dairy consumption – 141.5kg/year per capita (ABS 1992) and a high proportion of land and water dedicated to agriculture – 33 per cent of national total in 1975 (UNEP 1991; OECD 1994a).

Extracting such inputs causes significant environmental impacts, especially since European agricultural methods and animals are generally unsuitable for Australia's thin, nutrient-poor topsoil layer and extreme climatic conditions. The extraction (and frequent wastage) of non-renewable fossil groundwater sources from artesian basins is a common agricultural practice in Australia (Aplin 1999). Forest clearance, excessive grazing and irrigation, energy-intensive agricultural methods and the widespread application of fertilizers and pesticides are also linked to agricultural inputs into industrialized Australia. For instance, 3 871 000 metric tonnes of fertilizers were applied in Australia in 1990 over 27 360 000 hectares (FAO 1992a, 1992b) and an annual mean of 65 200 tonnes of pesticides were applied between 1982 and 1984 (UNEP 1991). Subsequent environmental impacts include soil erosion, desertification, soil salinity and waterlogging, native species extinction, dieback in trees, and eutrophication and toxic algal blooms in hydrological systems (Aplin 1999). Transportation of agricultural goods, food processing plants, cooking, freezing, canning and packaging are other material-intensive processes related to Australian agricultural inputs (see, for instance, Kranendonk and Bringezu 1993). Corporate-scale abattoirs, feedlots and battery hen factories are common in Australia and must be questioned on ethical grounds as well as in terms of environmental and human health implications (Aplin 1999). Hence the material flows, environmental impacts and ethical implications of modern Australian agricultural inputs are significant indeed.

Forestry inputs

Australia is a major exporter of woodchips (6.5 million tons/yr) and particles, and a major importer of sawnwood, fiberboard, newsprint, paper and paperboard (FAO 1993). Together with the agricultural industry, the forestry industry has been responsible for the widespread clearance of Australia's forests. In little over 200 years since white colonization, 95 per cent of Australia's forests and wetlands have been lost, with subsequent widespread extinction of many plant and animal species (WRI and UNEP 1994; OECD 1994a).

Most forests in Australia are logged by clearfell methods, resulting in extensive habitat loss for many native plant and animal species, soil erosion, slope instability, flooding and increased siltation of water systems. Massive areas of old growth forests with extremely valuable biodiversity values are still logged to make woodchips. Such forestry practices are clearly unsustainable. However, wood is one of the least energy-intensive and most renewable construction materials available. Plantations are more productive than native forests and, if managed prudently, could lead the way towards ecologically sustainable forestry (Aplin 1999).

Mineral inputs

Australia is the world's largest exporter of black coal, alumina, diamonds, ilmenite, rutile and zircon, the second largest exporter of iron ore, aluminum, lead and zinc, and the third largest exporter of gold (ABS 1994b; IEA 1999c). This illustrates the current importance of mineral inputs to the Australian economy.

Data on the consumption of minerals in Australia are quite insufficient. However, it is likely that most mineral inputs are used for increasing industrial production and urbanization (see, for example, Baccini and Brunner 1991). This extension of the anthroposphere is driven not so much by population growth as by the increasing demand for residential areas and roads typical of affluent societies.

Mining activities reduce future land productivity, and produce toxic wastes and large translocations of materials. Metal ore flows are especially harmful to the environment in qualitative terms. Australia's continuing expansion of coal mining (see, for example, NSWDMR 1998) also contradicts its greenhouse gas policies since coal burning is a significant contributor to global warming. Mining in Australia is also a politically sensitive issue since mining interests frequently conflict with Aboriginal land rights and the protection of sacred sites (Connell and Howitt 1991; Moody 1992). As is recognized internationally, uranium mining is particularly dangerous to humans and the environment, and radioactive leaks, spills and accidents are frequent at uranium mining sites (for example, Anderson 1998; CCNR 2000; WUH 1992). No technology exists to safely contain the radioactive waste (for example, Lenssen 1996; Mudd 2000; *The Ecologist* 1999). Despite strong protest, Australia's current Conservative Government is expanding uranium mining in Australia, including mining in the ecologically and culturally sensitive Kakadu National Park (Wasson *et al* 1998; UNESCO 1998; Gunjehmi Aboriginal Corporation 1998). A national industrial ecology dependent on uranium inputs adversely affects global industrial ecology since radioactive wastes are often exported and uranium may be used for nuclear proliferation (for example, Roberts 1995; Muller 1995; Skor 1998; UN 1996b). Hence Australia's industrial ecology mineral inputs involve significant social, cultural and environmental impacts, and inputs such as uranium must be seriously challenged.

Energy inputs

The major consumers of energy in Australia are the traditional industry sector, followed by the residential and transport sectors, as shown in Table 25.1. The most important energy commodity for Australia is coal, comprising 69 per cent of all the energy produced in 1992–3, and 78.8 per cent of the fuel used to produce electricity (IEA 1999c). The use of renewable energy sources such as solar energy and plantation wood is insignificant in comparison.

Energy consumption produces significant pollution and involves massive amounts of water consumption. Political tensions over access to traditional energy supplies such as oil already exist and are likely to increase as such supplies dwindle. Unfortunately, the Australian Commonwealth Government places low priority on developing renewable energy sources. Nevertheless, some efforts are being made to promote cogeneration in industry (ibid.). Methane gas from municipal waste is now fueling five power stations in the country, and wind farms have great potential for low-cost power supply (ABS 1994a). Solar energy is especially suitable for Australia's sunny climate. Australia has some of the world's most advanced solar cell technology. Currently, some 10000 Australian households generate their own electricity by solar energy and 5 per cent of Australian homes have solar water heaters (ibid.). Thus, although Australia's current energy inputs indicate

Table 25.1 Final consumption of energy fuels by sector in Australia, 1992 (1000 metric tonnes, unless otherwise specified)

Type of fuel	Industry	Transport	Agriculture	Commercial & public service	Residential	Total
Steam coal	2525		—	120	7	2652
Sub-bit. coal	2519	193	—	1	—	2713
Lignite	53	—	—	—	—	53
Oven & gas coke	296	—	—	—	—	296
Pat. fuel & BKB	386	—	—	71	5	462
Natural gas (TJ)	289102	5733	50	36278	91867	423030
Gas works (TJ)	5385	—	—	683	1585	7653
Coke ovens (TJ)	25898	—	—	—	—	25898
Blast furnaces (TJ)	27123	—	—	—	—	27123
Electricity (GWh)	60478	1885	2564	28855	40333	134115
Crude oil	3	—	—	—	—	3
Refinery gas	8	—	—	—	—	8
LPG & ethane	956	515	18	153	174	1816
Motor gasoline	—	12461	—	—	—	12461
Aviation gasoline	—	70	—	—	—	70
Jet fuel	—	2735		—	—	2735
Kerosene	9	—	3	14	132	158
Gas/diesel	1495	4955	1057	57	50	7614
Residual fuel oil	869	350	—	20	—	1239
Naphtha	199	—	—	—	—	199
Petrol. coke	32	—	476	—	—	508
Other prod.	13	—	2203	—	32	2248
TOTAL	417349	28897	6371	66252	134185	653054

Source: IEA (1999c).

an unsustainable industrial ecology, there is significant potential to shift this trend towards smaller-scale, efficient and renewable energy inputs.

Transport inputs

Australia's transport industry relies heavily on motor vehicles, passenger cars in particular, as is shown in Table 25.2. The 'Australian Dream' of having a large house and garden per household, together with the poor urban planning of Australia's colonial history, has resulted in extensive urban sprawling (Spearitt and DeMarco 1988) which in turn has aggravated car dependence. Such transport inputs are unsustainable because of their high and inefficient uses of energy and land, their high air and noise pollution, and the high accident risk associated with motor car use (Stiller 1993). The increasing demand for larger (6–8-cylinder) cars and multiple car ownership per household in Australia only exacerbates these unsustainable trends (IEA 1999c). By the same token, Australia's transport structure offers significant potential to close the materials cycle and reduce wastage of resources since it is responsible for a very large share of the nation's material and energy inputs and outputs. (See Enquête 1995, p.28, for a discussion of excellent transport policy

Table 25.2 Transport characteristics in Australia

Roads	
Network length (km), 1991	
All roads	853 000
Motorways	1 100 000
Vehicle stocks, 1991	
Motor vehicles in use	10 002 000
Passenger cars in use	7 850 000
Goods vehicles in use	2 150 000
Traffic volumes (billion vehicles/km), 1991	
Total vehicles	161.6
Passenger cars	124.9
Goods vehicles	36.4
Car ownership (vehicles per 100 people), 1991	45
Number of registered vehicles, 1991	10 505 900
Distance traveled (million km in a year), 1993	151 154
Purpose of travel, 1991	
Business purposes	34.8%
Travel to & from work	22.5%
Private purposes	42.7%
Rail	
Government railway passenger journeys (1000), 1992–3	
Suburban	393 088
Country	8 306
Total	301 394
Air	
Air traffic to Australia, 1992–3	
Number of flights	26 207
Number of passengers	4 902 693
Air traffic from Australia, 1992–3	
Number of flights	26 088
Number of passengers	4 855 572

Sources: OECD (1993a, 1994a), ABS (1994b).

strategies which could facilitate decreasing the material intensity and associated social, economic and environmental costs of Australia's transport system.)

Australian Industrial Ecology Outputs

Outputs are here defined as those materials flowing from the industrial system to the consumers and finally to the environment. Sources of outputs can be point sources, such as specific industrial plants, or *diffuse* sources – producing outputs from the dissipative use of materials (Ayres *et al.* 1987). The major outputs from Australian industry to the environment considered here are air, toxic and urban solid waste outputs. They are summarized for various years in Table 25.3.

Table 25.3 Waste generation in Australia

Waste type	Date	Amount (1000 MT)
Hazardous waste	Early 1990s	316
Exported		0.7
Industrial wastes dumped at sea	1985	425
Municipal wastes	1979	
Landfill		9 800
Incineration		200
Solid waste		
Municipal	1980	10 000
Industrial	1980	20 000
Sewage sludge	1982	45

Sources: OECD (1994a), UNEP (1991).

Australian waste data are not available in as much detail as for the USA and some European countries, but efforts are being made to improve waste classification, data collection and reporting systems across Australia (Moore and Tu 1995).

Outputs to air

Australia's per capita emission of greenhouse gases are among the highest in the world, reflecting the dominance of coal and motor cars in Australia's energy and transport sectors, respectively (see Table 25.4). The continued destruction of the ozone layer is also a significant environmental impact of Australian industrial systems both nationally and internationally (Brown and Singer 1996, pp. 7–16).

Table 25.4 Greenhouse gas emissions in Australia from anthropogenic sources, 1991

Emission type	Emissions (1000 MT)	Emission source	Emissions (1000 MT)
Carbon dioxide	149 557	Carbon dioxide	
Solid	149 557	Mobile sources	66 700
Liquid	76 644	Energy transf.	145 900
Gas	32 258	Industry	46 000
Cement manuf.	3 363	Other	12 000
Total CO_2	261 818	Methane	
Total CO_2 per capita	15.1	Livestock	570
Methane	4 500	Coal mining	1 400
Chlorofluorocarbons (CFCs)	5 000	Solid waste	330
		Oil & gas production	60
		Wet rice production	2 100

Sources: OECD (1993a); WRI and UNEP (1991).

Toxic outputs

Australia ranks 12th in heavy metal exposure and 13th highest in human risk exposure in the world in terms of toxic outputs released to the environment from industrial activities (WRI and UNEP 1994).

Urban solid waste

Sydney urban solid waste data indicate data typical of urban Australia. Currently, the annual disposal rate of urban solid waste in Sydney is about 3.4 million tons and rising rapidly (WMA 1990). The vast majority of urban solid waste in Australia is disposed of in landfills (89 per cent and 100 per cent in New South Wales and Victoria, respectively, in 1994; Moore and Tu 1996). Owing to the crisis in finding landfill space, New South Wales has recently prioritized minimizing waste production (NSWPMB 1998).

Municipal household or domestic solid waste comprise most of the urban solid waste produced in Australia (42 per cent in 1994). The vast majority of domestic solid waste components, such as paper, organic compostable, plastic and glass, can all be re-used or effectively recycled (WMA 1990). Even smaller volume components such as household hazardous wastes, ferrous wastes and non-ferrous waste have ecologically sound alternatives or can be recycled very successfully, thus reducing the need for mining virgin materials (Ayres and Ayres 1999b; WMA 1990). However, Australia's recycling rate is very low despite the ready availability of suitable materials, technology and public demand (WMA 1990; Moore and Tu 1996).

The main manufacturing industry contributors to manifested hazardous waste in Sydney are chemical, petroleum and coal products, basic metal products, fabricated metal products and miscellaneous manufacturing. Waste generation is increasing for these industries (Moore and Tu 1996). Australia also exports hazardous wastes and dumps industrial wastes at sea (UNEP 1991), thus adversely affecting global industrial ecology. Identifying these sources and flows of human-induced waste outputs is a vital step in identifying strategies to improve Australia's industrial ecology.

Paths of material flows in Australia

The processes which transform inputs from the environment into outputs to the environment have been described briefly above in both qualitative and quantitative terms. Processes such as resource extraction, manufacturing, packaging, transport, recycling and disposal are all important in determining which paths material flows take. Such flows indicate the level of material throughput that is occurring within Australia's industrial ecology.

Materials Stocked in the Anthroposphere

Residential buildings and length of roads give some rough guide to the level of materials stocked in Australia's anthroposphere. A significant amount of materials are stocked in residential buildings, with detached houses (76.7 per cent) being the most common dwelling types in Australia (ABS 1994b). Separate houses have a particularly high material intensity since most households have three bedrooms (48.7 per cent), garage one or two vehicles (72.9 per cent) and use brick as their outer wall materials (86 per cent) (ibid.).

Australia's National Material Intensity

Unfortunately, data are not available to calculate accurately Australia's 'total material consumption' and 'total material emissions', and thus its national material intensity. However, the above sections on inputs, outputs and materials stocked provide a preliminary indication that Australia's current industrial ecology is highly material-intensive. Inefficient levels of recycling, waste mining, dematerialization and dissipative resource use control also inhibit closing of the materials cycle, thus hindering the development of a sustainable Australian industrial ecology.

Possible Key Forces Driving Australian Material Flows

Australia has unique climatic, topographical, situational and anthropological characteristics compared with Europe and North American, and thus assumptions and strategies used to improve industrial ecology elsewhere are not necessarily appropriate for Australia. Outlining Australia's unique characteristics may help identify possible key forces driving Australia's material flows.

Climate and topography

Australia covers 7682 thousand square kilometers and has an abundance of natural resources, including some of the largest internationally important wetlands, major protected areas and world heritage sites in the world, as shown in Table 25.5. Australia is the lowest, flattest and second driest continent in the world, with rainfall (or the lack of it!) being the single most important factor affecting land use and rural production (ABS 1994b). Australia's hydrology is unique, dominated by low-frequency, high-magnitude floods (Brierley 1995). Droughts are common and sunshine is abundant. Frequent bushfires play an integral role in the regeneration of Australia's forest areas and biodiversity. During the last glacial retreat Australia's landscape was not reworked, as occurred in the northern hemisphere (Aplin 1999). Australia's climate and geomorphology are thus very different from those of many other industrialized countries and consequently require locally appropriate strategies to promote a sustainable national industrial ecology.

Table 25.5 Natural resources in Australia, 1990

Type of resource	Extent of resource	World rank of extent
Biosphere reserves	12 sites, 47432 sq.km	—
Wetlands of international importance	39 sites, 44779 sq.km	3rd highest
Major protected areas:		—
Scientific reserve	16 sites	4th highest
National parks	339 sites	highest
Nature reserves	309 sites	2nd highest
Protected landscapes	64 sites	2nd highest
Total	728 sites, 456500 sq.km (5% of total territory)	2nd highest
World heritage sites	8 sites	2nd highest

Source: OECD (1993a), UNEP (1991).

Population and urban structure

Australia is well recognized for its low population and population density (17 529 000 inhabitants and 2.3 inhabitants per square kilometer, respectively, in 1992; ABS 1994b). Australia also has an aging population, with population growth due not so much to domestic birth rates as to high immigration rates (ABS 1994b). Australia has one of the highest proportions of urban population in the world (85 per cent in 1990) and urban growth remains steady (UNEP 1991). As discussed above, Australia's urban structure is highly resource-intensive. Hahn (1991) offers some excellent ecological urban restructuring strategies which could be applied, with local variations, to help dematerialize Australia's urban structure.

Political/legal characteristics

Australia has three levels of government – federal, state and local – and interests, legislation and responsibilities between and within the various government levels frequently conflict (Boardman 1990; Toner and Doern 1994). For instance, Australia has endorsed national strategies for ecologically sustainable development and for reducing greenhouse gases, but federal government's main focus remains on increasing Australia's competitiveness in energy production internationally (ABS 1994b; IEA 1999c). A comprehensive national integrated resource management strategy would greatly assist Australia's transition to sustainable industrial ecology and overcome the currently fragmented legislation (NCC 1999). Thus Australia's particular political/legal structure is a significant force driving national material flows, and must be considered when canvassing strategies to improve Australia's industrial ecology.

Public opinion and will to act

The most powerful pressure forcing governments and industry alike to improve Australia's industrial ecology is public pressure (Jänicke and Weidner 1995; Enquête 1994). Public concern for the environment has been widespread in Australia, as indicated by sustained and widespread campaigns to prevent damming, mining and logging of ecologically sensitive areas, as well as popular support for recycling and energy efficient technology. Public support for the environment is also indicated by the growing importance of the Australian Greens as a political party (Brown and Singer 1996).

There are, however, many Australians who pursue environmentally destructive goals of having bigger houses, bigger cars and more motorways (IEA 1999c). Structural factors, such as the development of car-dependent urbanization, as well as Australia's abundant natural resources and low population density, may partly disguise and encourage such unsustainable behavior (Boardman 1990). Raising and harnessing public awareness of the need for a sustainable industrial ecology in Australia is therefore a high priority when considering anthropological forces driving material flows.

Economic characteristics

As discussed earlier, Australia is a major energy and mineral-exporting country and Australia's national economy is influenced by competition within the international economic structure (Bryan and Rafferty 1999; Wiseman 1998). However, it is interesting to note that the manufacturing, electricity, mining, and gas and water industries have a decreasing role in employment, and that more people are now working in the service sector (ABS 1994a). Similarly, GDP contributions are decreasing for the manufacturing, construction, agriculture, forestry and fishing industries (ABS 1994b), possibly indicating a restructuring of industry away from 'dirty' industries to the potentially less energy and material-intensive service industries (Simonis 1994; Jänicke, Mönsch and Binder 1994). Although much of Australia's economy is based on promoting resource extraction industries, the scope exists to restructure industry to meet employment and economic needs while simultaneously protecting the environment (Brown and Singer 1996, pp. 119–53).

CONCLUSIONS AND FUTURE DIRECTIONS OF AUSTRALIA'S INDUSTRIAL ECOLOGY

Industrial ecology is an internationally recognized material accounting procedure useful for providing a comprehensive model of significant material and energy flows both within industrial systems and between industry and the environment. The industrial ecology approach is especially significant in that it can provide information currently lacking in most material accounting procedures, including the following:

- cumulative effects of material flows;
- estimates of historical and future flow patterns;
- diffuse sources of material output flows to the environment;
- possible political, economic, technological, social and other forces driving human-induced material flows.

The Australian case study usefully outlines industrial ecology methodology and illustrates the need to consider how unique environmental and human factors influence material flows within a region when attempting to improve that region's industrial ecology. National case studies also play an important role in contributing towards a globally sustainable industrial ecology, since all nations are interlinked economically, politically and environmentally.

This case study provides a preliminary insight into Australia's industrial ecology as highly material-intensive, but also identifies many opportunities to improve its sustainability. For instance, Australia's transport structure offers great potential to improve the nation's industrial ecology, since it is currently consumes a very large share of national material and energy resources. For more details on possible political, economic, technological, informational and social instruments to improve Australia's industrial ecology, see Durney (1997) and Brown and Singer (1996).

A key finding of the Australian case study was that, compared to the USA and some European countries, Australia has a paucity of data on major inputs, outputs and paths of material flows. Further, all countries have the following data priorities to improve their industrial ecology studies and evaluations:

- data on the output side of material flows, particularly cumulative, dissipative, hazardous and toxic wastes and their long-term effects (for example, Baccini and Brunner 1991);

- the composition of goods: supplementing financial bookkeeping with material bookkeeping (Stigliani and Anderberg 1994);
- material consumption data for industrial activities (Liedtke 1993);
- emission coefficients for consumption activities and post-disposal impacts (Ayres and Ayres 1994);
- data on ecological 'rucksacks' and translocated masses, both nationally and for exports and imports (Bringezu *et al.* 1994: Schmidt-Bleek 1993a).

It would also be important to consider trade and international relations and equity, employment and distributional issues when trying to facilitate improved national industrial ecology, and to guard against imperialist and non-participatory strategies to improve national and global industrial ecology. See Gerd *et al.* (1989) on exports and dematerialization; Dietz and van der Straaten (1994) and Simonis (1992) on distributional effects of dematerialization; Godlewska and Smith (1994), Durning (1994) and Escobar (1985) on imperialism in 'development'; and Howitt (1993) and Lane (1997) on participatory strategies of assessing social impacts of developments.

In Australia major efforts are being made to improve the national waste database (Moore and Tu 1995, 1996), to facilitate integrated resource management in Australia (NCC 1999) and to build an environmental impact database for various activities (P. Hopper, Nature Conservation Council, Sydney, personal communication, July 2000). Building on these strengths while incorporating useful international industrial ecology research can improve Australian industrial ecology methodology, and identify policy instruments to help close and dematerialize Australian industrial material cycles. Modeling national industrial systems on sustainable ecological systems with low material intensity and throughput is a vital step towards the broader goal of a global sustainable industrial ecology.

26. Industrial ecology: the UK

Heinz Schandl and Niels Schulz

Industrial ecology aims at an ecological restructuring of the industrial economy, fostering environmental soundness in production and consumption. It has been argued that this aim could be supported by positive side-effects of structural change (Jänicke *et al.* 1989), leading to economic and ecological advantage at the same time. This has also been referred to by the notion of an 'efficiency revolution' (Weizsäcker *et al.* 1997) or 'dematerialization'. Structural change and technological innovations can be either supported or hindered by political interventions aimed at changing the framing conditions of industrial activities.

An integrated economic and environmental policy can provide such a framework and thus intervene in the economic process to support ecological improvements within the economy. Such an approach would profit by a thorough understanding of the system dynamics of society's interaction with ecosystems. One mode of this interaction is society's industrial metabolism. Understanding of the characteristics of this metabolism, both historically and currently, supports our understanding of the functioning of complex society–nature interactions and helps to increase the chance of successful interventions aimed at future ecological modernization of the production system (Christoff 1998).

Indicators derived from an accounting on the basis of the theoretical concept of industrial metabolism help to reduce complexity and thereby to move decision-making processes in the direction of sustainability. These indicators should be theoretically correct, policy-relevant and feasible. The material flow accounting approach offers a complementary accounting framework that can generate useful indicators for policy makers (for methodological details, see Chapters 8 and 11).

In this chapter we discuss the material aspect of industrial metabolism for the UK's economy in a historical perspective, since current metabolic patterns are rooted in past developments. Sharp breaks in trends have been taken into consideration. See, for example, Sieferle's discussion of energy (1982). The first section briefly refers to examples of an industrial ecology approach in the UK context. We then apply material accounting mainly to material inputs to the UK economy, taking exports into consideration. Time series for the material inputs to the UK economy from 1937 to 1997 are shown and discussed. This data set has been established on the basis of periodically available official sources following a 'top-down' approach. At some points we had to rely on estimations such as for animal grazing or for timber harvest before 1970. The physical data set was cross-analyzed with macroeconomic indicators such as GDP and then compared to an average metabolic profile of industrial economies derived from previous studies.

INDUSTRIAL ECOLOGY IN THE CONTEXT OF THE UK

Industrial ecology depends more on physical than on monetary environmental accounting. If we consider the UK in comparison to other European countries, most past efforts have concentrated on economic approaches, such as correcting the SNA (System of National Accounts) for environmental side-effects within an SEEA (System of Integrated Environmental and Economic Accounts) framework, or providing environmental input–output tables (again monetary). This remarkable neglect of the physical approach might be due to a long tradition of economic history (not only in the UK) which has always portrayed economic developments in monetary terms and rarely in physical terms. Nevertheless, there are some exceptions, both historic and recent.

Wrigley (1962) has contributed a picture of the physical dimension of the industrial explosion in the UK with respect to the supply of raw materials. Features of energy metabolism have been applied to explain historical processes in the UK by Richard N. Adams (1982). Brian R. Mitchell has published a compendium of socioeconomic data going back to the 19th century, covering several physical aspects (Mitchell 1988). The UK *Annual Abstract of Statistics* (published since the 1850s) is also a reliable and valuable data source for a time series approach.

Recently, the UK Office for National Statistics' work on environmental accounts began to include a physical accounting for the UK's foreign trade activities (Vaze 1998). Further, at the Manchester School of Geography, the minerals and fossil fuels fraction of the UK industrial metabolism has been linked to geomorphological, environmental and land use change (Douglas and Lawson 2000). The Manchester approach explicitly deals with mobilized materials not intended to enter the economic process, including overburden from mining or translocated materials. These large hidden flows in industrial metabolism, consisting of mainly trouble-free materials (see Steurer 1996) are discussed in comparison to flows which are mobilized by ecosystems dynamics and recognized as problems from the point of view of sustainability. Furthermore, city metabolism for Manchester has been studied following a historical perspective (Douglas, Hodgson and Lawson forthcoming, see also Chapter 28). Projects with a more technical, regional or sectoral focus or with a focus on specific materials (such as heavy metals) have been linked together in an initiative under the heading 'Mass Balance Club'. Berkhout (1998) has contributed to critical secondary analysis of aggregate resource efficiency and to the question of policy relevance of environmental indicators.

Notwithstanding as reference articles indicate (Fischer-Kowalski and Hüttler 1999; Cleveland and Ruth 1999; see also Chapters 2 and 12) there has been no attempt to provide an economy-wide MFA data set for the UK so far. To partially overcome this lack, we present recent work done at the Institute for Social and Economic Research at the University of Essex in more detail hereafter.

AN EMPIRICAL APPLICATION FOR THE UK ECONOMY

The data set for material inputs into the UK economy has just recently been established. For a complete description of the methodological approach and the data sources, see Schandl and Schulz (2000). The account was established on the basis of yearly available data sources for the UK, such as agricultural statistics, statistics on supply and demand of home-grown timber, the UK's mineral statistics, the overseas trade statistics and the input–output tables. Some first insights on the UK's physical economy can be drawn from time series analysis. At an aggregate level we distinguish biomass (plant harvest, timber removals, fishing and hunting) from mineral materials (ores, industrial materials and construction materials), fossil fuels (coal, crude oil, natural gas) and products (both semi-manufactured and finished). At this level, water and air inputs are not included.

Within the MFA accounting framework water appears as a direct input (processing water) and as the water content of materials. Careful accounting for water is especially necessary when balancing materials. Similarly, oxygen and nitrogen in the air are direct inputs due to conversion processes such as incineration or production of cement and fertilizers. In our account we treat all materials with the water content when marketed. Exceptions are made for biomass for grazing, which is included with a standardized water content of 14 per cent, and timber, which is included using the water content when removed from the forest.

Looking at the physical dimension of the UK economy within the last decades clearly shows that (after a period of rapid growth from the 1940s to the 1970s) material input had come to a standstill. Direct material input (DMI) is one of the internationally agreed indicators derived from a material flow accounting approach. Direct input consists of mainly economically used materials from domestic material extraction and imports. In the 1940s, average DMI accounted for 413 million tons (or 8.5 tons per capita). The Fordist compromise between capital and labor (Boyer 1979; de Vroey 1984) led to a new regime of accumulation and went hand in hand with a specific metabolic regime characterized mainly by rapid growth of yearly inputs of minerals and fossil fuels. As a result a new level of overall material input at an average of 774 million tons was reached in the 1970s. While minerals (ores, construction minerals and industrial minerals) and fossil fuels (coal, crude oil and natural gas) input grew massively (140 per cent growth for minerals, 46 per cent growth for fossil fuels between the 1940s and the 1970s), biomass input remained more or less stable (only 18 per cent growth between the 1940s and the 1970s). Products, both semi-final products and final goods, show a different pattern and seem to be more vulnerable to change than other inputs (see Table 26.1). Fluctuations in imported products are mainly due to changes in semi-manufactured materials serving as intermediate materials to establish the production infrastructure. This group of materials is more vulnerable to economic fluctuations than consumer goods are (see Hunt and Sherman 1972).

Table 26.1 Yearly average materials input to the UK economy over six decades (MMT)

	1940s	1950s	1960s	1970s	1980s	1990s
Biomass	79.6	84.5	89.5	91.9	95.1	97.7
Minerals	115.0	194.9	300.5	361.5	318.4	317.1
Fossil fuels	203.3	261.1	270.6	304.3	326.8	311.3
Products	15.6	6.2	9.6	16.3	32.2	51.0
DMI	413.4	546.7	670.2	774.0	772.5	777.2

Source: Authors' calculation from UK Office of National Statistics, UK Forestry Commission, British Geological Survey and UK Department of Trade and Industry.

In the 1970s there was a turning point both in the economic and the industrial metabolic regimes. First of all, overall growth came to a standstill. The UK economy has stabilized the material inputs at a high level. Nevertheless, different aggregates show a different trend. While fossil fuels and imported products still contributed to growth, there had been a comparably sharp decline in mineral materials, especially from the 1970s to the 1980s. At the same time, biomass input to the UK economy remained a more or less stable fraction of socioeconomic industrial metabolism, though growth rates also point downwards (see Table 26.2).

Table 26.2 Relative change of average materials input to the UK economy (per cent)

	Average 1940 to average 1950	Average 1950 to average 1960	Average 1960 to average 1970	Average 1970 to average 1980	Average 1980 to average 1990
Biomass	6.2	6.0	2.7	3.4	2.8
Minerals	69.5	54.2	20.3	−11.9	−0.4
Fossil fuels	28.4	3.6	12.5	7.4	−4.7
Products	−60.0	54.2	68.9	98.4	58.3
DMI	32.2	22.6	15.5	−0.2	0.6

Source: Authors' calculation from UK Office of National Statistics, UK Forestry Commission, British Geological Survey and UK Department of Trade and Industry.

DOMESTIC EXTRACTION OF MATERIALS

What follows is a more detailed description of the different input aggregates, domestic extraction and imports. Biomass extraction (plant harvest, fishing and timber removals) is the most stable part of the UK industrial metabolism over time. A closer look shows that agricultural crop mix has undergone considerable change since the early 1960s. The yearly amounts of harvested cereals and fodder crops were raised after World War II, owing to changes in agricultural land use patterns and intensification processes. Intensification on agricultural land in the 20th century involved the replacement of animal traction by machine traction, a more intensive use of chemical fertilizers and pesticides and, finally, the introduction of genetic alteration of plants. With the final step of industrialization in agriculture, the level of crop harvest increased by one-third (see Table 26.3).

Timber figures are still too preliminary to justify a final comment. However, since 1970–74, timber removals from UK woodlands steadily increased, from 2.1 million tons to 5.8 million tons in 1997. This is partially due to land use changes in the UK since 1937, when woodland only accounted for 5.1 per cent of total area. As a result of afforestation efforts the proportion of woodland has increased to 10.9 per cent of total land area. But the mix of trees, with an increasing proportion of conifers (now 82 per cent of all woodland), suggests a clear trend to monoculture.

The domestic extraction of mineral materials is a different story. Iron ore and other ore extraction, being closely linked to the industrial–military production process, did not show the typical decline or stagnation during wartime. The period 1940–44 experienced

Table 26.3 Average domestic extraction of materials for five-year periods in the UK, 1937–97 (MMT)

Period	Cereals & fodder crops	Other biomass harvest*	Timber	Iron ore & other ores	Industrial minerals	Clay	Sand & gravel	Crushed stone	Coal	Natural gas	Crude oil
1937–39	18.96	27.08	3.64	13.54	16.70	30.09	25.96	31.74	232.91		0.13
1940–44	25.02	33.16	3.47	18.11	15.74	16.32	27.74	30.61	205.45		0.20
1945–49	23.62	35.46	3.22	12.79	16.93	19.92	28.37	29.49	198.96		0.16
1950–54	23.78	34.31	3.63	15.28	27.90	31.14	46.70	41.77	225.10	0.01	0.16
1955–59	20.66	32.65	3.84	16.02	33.04	32.47	63.92	50.05	221.88	0.05	0.15
1960–64	21.03	35.67	3.43	16.29	37.85	35.58	89.62	67.08	198.49	0.17	0.14
1965–69	24.34	37.00	3.05	13.75	39.36	38.92	112.26	108.43	174.72	2.49	0.08
1970–74	31.86	35.04	2.15	8.42	35.92	35.11	122.82	142.18	132.12	35.97	0.09
1975–79	31.40	34.70	2.68	4.28	36.66	26.46	114.19	128.89	124.14	57.43	36.77
1980–84	36.04	37.68	3.19	0.57	28.01	20.86	101.89	112.31	109.92	50.51	100.61
1985–89	35.20	36.95	4.14	0.19	25.00	19.41	120.16	146.21	97.03	33.52	123.99
1990–94	31.70	37.55	4.75	0.01	17.98	11.80	102.22	156.47	75.14	44.05	94.13
1995–97	32.18	38.18	5.76	0.00	18.21	12.88	92.03	144.68	50.91	64.94	129.50

Note: *This category includes potatoes, sugar beets, fruits, vegetables, straw and hay, and biomass from fishing and animal grazing.

Source: Authors' calculation from UK Office of National Statistics, UK Forestry Commission, British Geological Survey and UK Department of Trade and Industry.

the highest domestic iron ore extraction ever, 18.1 million tons per year. Nevertheless, the UK mining sector faced strong declines from the mid-1960s on and nearly closed, with only around 0.6 million tons ores extracted yearly since 1980–85.

Clay extraction (for the production of bricks) was traditionally high in the UK, around 30 million tons yearly until the 1970s. A period of sharp decline followed. In recent years (1995–7), clay extraction was 12.9 million tons yearly. The aggregate of minerals for industrial use is similar to that of clay. In the late 1990s, industrial mineral extraction was around 14.7 million tons yearly. More or less the same trend is seen in gypsum and anhydrite, with salt at a still lower level.

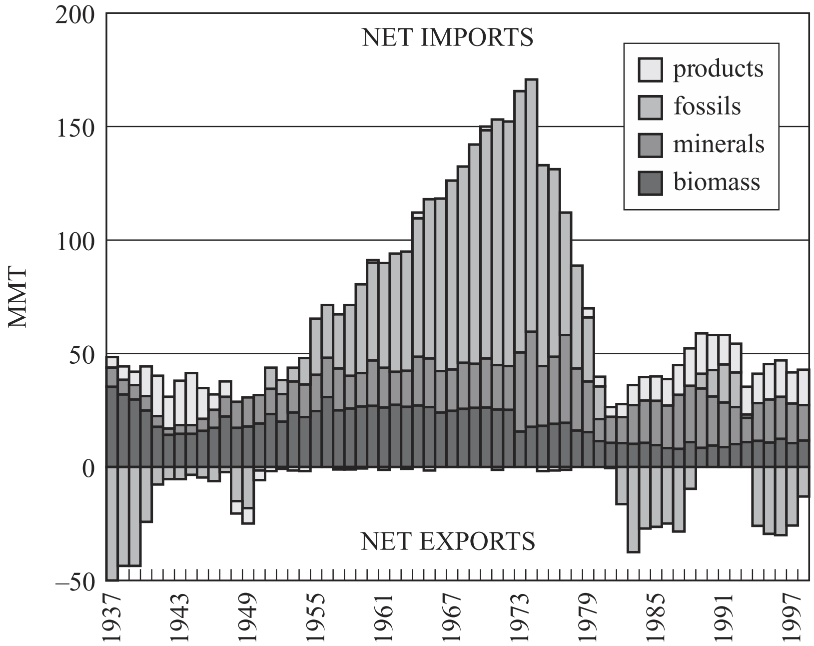
For massive minerals used in construction activities (sand, gravel and crushed stone) the story is similar. Starting at a level of 30 million tons before and during the war, the domestic use of construction minerals grew explosively to reach 140 million tons of crushed stone and 120 million tons of sand and gravel around 1975, four times the original level. This was followed by a decade of sharp decline up to 1985 and then a short phase of resurgence, when both aggregates reached the level of 1975 in the period 1985–90. From then on, use of crushed stone and of sand and gravel, which were previously linked, moved in different directions. Crushed stone stabilized around 145 million tons yearly, whereas sand and gravel consumption has recently fallen to around 95 million tons. This is mainly due to new extraction sites for crushed stones on the Scottish coast, which largely produce for export.

Fossil fuels extraction has been a dominant feature of the UK industrial landscape. Here we must distinguish between two phases, the coal era and the more recent regime when natural gas and crude oil started to play a major role and finally replaced coal to a large extent. Coal was the main energy source for the UK economy until 1960. What followed has been a journey on a roller coaster. Starting between 1955 and 1959, the descent began, somewhat cushioned since 1970–74, but steady. Currently, coal output amounts to 50 million tons per year. Clearly, production declined even before domestic (North Sea) extraction of natural gas and crude oil became important. Natural gas extraction started in 1968. Crude oil pumping began on a large scale in 1976, shortly after the first oil crisis.

Domestic (UK) fossil fuel output began a resurgence in 1975, which lasted until 1983 (see Table 26.3). The year 1984 experienced a sharp decline due to miners' strikes. Although output recovered somewhat, the late 1980s and also the early 1990s were periods of further decline, the lowest level occurring in 1994 (201 million tons). In the year 1997, domestic extraction of all fossil fuels was 249 million tons.

MATERIAL FLOWS FROM FOREIGN TRADE

The second part of society's industrial metabolism stems from the interaction with other economies. In the UK case, not surprisingly, fossil fuels are also the main component of imports and exports. Imports of fossil fuels became relevant after World War II and reached a peak during 1970–74, when around 130 million tons (mainly crude oil) were imported yearly. Up to this point, fossil fuels also made a dominant contribution to the UK exports, mainly as coal. This changed with the start of the North Sea gas and oil industry. Since around 1975, imports of fossil fuels have decreased considerably while exports have exploded. If we look at the foreign trade with fossil fuels from the perspective of a physical net balance of trade, the UK began to be a net importer of fossil fuels in the 1950s, reaching a maximum of 115.5 million tons import surplus. But 1973, the year of the first oil crisis, seems to be a turning point. By 1981, the UK was a net exporter of fossil energy carriers. Except for 1988–94, this has continued. For all other materials, the UK economy relies mainly on imports from other countries (see Figure 26.1). Interestingly, Vaze *et al.* (1998) argue that the dependence of the UK economy on foreign resources is predominantly satisfied through trade relations with other European countries.



Source: Author's calculations from UK Office for National Statistics and UK Department of Trade and Industry.

Figure 26.1 A physical net balance of foreign trade activities for the UK economy for the period 1937–97 (MMT)

Main imports are biomass and industrial minerals. For biomass, the foreign trade balance stabilized at 25 million tons net imports yearly through to 1973. This balance has improved somewhat since the 1970s, when biomass exports from the UK increased significantly. The import surplus for mineral materials was largest between 1973 and 1979 and again in the late 1980s. The early 1980s saw some structural change in UK industrial activities and subsequently led to lower rates of mineral demand. Since around 1985, overall imports have dominated exports by 20 million net tons yearly.

Interestingly, the segment of imported semi-finished or final products (both for intermediate use and final consumption) also contributed to net imports between 1937 and 1948 and again from the 1980s on, or at least balanced. A decomposition of products for intermediate use and final consumption might tell a slightly different story. Nevertheless, the UK clearly is no longer the workshop of the world. Industrial production has sharply declined since the 1980s, which explains the surplus of imported products.

Since 1937, the UK has been an economy showing a positive balance of trade in monetary terms. However, this was not reflected in the physical balance of trade.

DEMATERIALIZATION

So far we have only discussed developments in aggregate material flows, mostly on a descriptive level. Nevertheless, it is also important to understand the relationship of materials input to other macroeconomic parameters such as population and economic development. At first sight, overall GDP seems to explain developments in direct material input, whereas population, being a rather stable factor, plays only a minor role. With regard to GDP, growth rates over the decades slowed down. It seems that the postwar institutional growth constellation was no longer available in the 1980s and subsequently economic growth rates stabilized, which also affected the material regime (see Tables 26.4 and 26.5). The situation is even clearer when we consider developments in industrial GDP compared to material flows.

Table 26.4 DMI per capita, GDP and population in the UK over six decades (per cent)

	1940s	1950s	1960s	1970s	1980s	1990s
DMI (tons per capita)	8.42	10.73	12.42	13.87	13.63	13.33
Real GDP (£ billion)*		267.71	358.77	466.01	562.23	689.13
Population (millions)	49.07	50.99	53.97	55.80	56.69	58.30

Note: * Corrected GDP at 1995 prices.

Source: Authors' calculations from UK Office for National Statistics.

Table 26.5 Relative change in DMI per capita, GDP and population in the UK over five decades (per cent)

	Average 1940 to average 1950	Average 1950 to average 1960	Average 1960 to average 1970	Average 1970 to average 1980	Average 1980 to average 1990
DMI per capita	27.4	15.8	11.7	−1.8	−2.2
Real GDP		34.0	29.9	20.6	22.6
Population	3.9	5.9	3.4	1.6	2.8

Source: Authors' calculations from UK Office for National Statistics.

On the other hand, population growth is a feature of the industrial society that is often underestimated. After all, the UK population grew from an average of 49 million in the 1940s to a current average of 58 million, in other words by 18 per cent. In contrast to GDP and DMI, population has undergone moderate change. Clearly, further investigation is necessary to gain an in-depth picture of the relation between economic variables, distribution variables and material flows.[1]

Apparent gains in UK resource use efficiency might well be a story of structural change and, to a lesser extent, technological advancement, rather than of a successful environmental policy. This argument is supported by the fact that, since the 1980s, coal mining and other traditional manufacturing activities have closed down and the UK economy has undergone a considerable transformation: from an industrial economy to a service economy. This probably had a positive side-effect for the environmental performance of the UK with respect to resource use. Several articles in *New Left Review* during this period attempted to describe the process of deindustrialization in the UK under the Thatcher administration (for example, Brenner 1998).

CHARACTERISTIC METABOLIC PROFILE OF SOCIETIES: THE POST-INDUSTRIAL PATTERN

Since the early 1990s, a number of empirical country studies estimating the resource basis of industrial economies have been developed (see Adriaanse *et al.* 1997; Schandl *et al.* 2000; Matthews *et al.* 2000). On the basis of these studies, some characteristics of industrial metabolism at the national level can be identified. One obvious feature of industrial metabolism is the enormous amount of throughput as compared to final output. This is true in the historical comparison with agricultural societies but also compared to recent industrializing societies and societies in transition.

Looked at more closely, the continuing high level of materials use appears to be a result of construction intensity, nutrition habits, energy supply and transport. The amount of consumer goods plays a comparably less important role, even though an enormous amount of resources, both materials and energy, are mobilized to produce them. The amount of physical advance achievements necessary to make available the production infrastructure and all payments due to the transport infrastructure to distribute final goods and the whole commerce infrastructure are also factors. Nevertheless, the dimensions of the material relations have changed dramatically. The metabolic profile of industrial societies is dominated by a small number of materials. Water, for instance, accounts for around 87 per cent of yearly mass throughputs in industrial economies. Air is approximately 8 per cent, whereas all the other materials (biomass, minerals, fossil fuels and imported products) only amount to around 5 per cent (Schandl *et al.* 1999). Even within the remainder, some materials dominate (for example, sand, gravel, crushed stone and rocks, fossil fuels, wood and feedstuffs for animals).

A second new feature in industrial metabolism is the growth dynamic, which is different from the agrarian mode of production not only quantitatively but also qualitatively. Whereas in agrarian societies production is limited by land availability and by the solar energy system, industrial society seems to possess limitless energetic resources. A further feature of the system is its low capacity for recycling. Currently, much less than 10 per cent of yearly throughput, outside of water and air, are kept within the recycling loops. It is even doubtful that the recycling potential can be raised significantly owing to the fact that many materials (such as fuels) cannot be recycled at all.

As has been argued before, industrial economies tend to use materials for a certain time period. These materials make up society's material components or, in other words, the material stocks. As a result mainly of construction activities, means of production and durable consumer goods net addition to stocks are relatively high. They amount to between 5 and 11.5 tons per capita and year (Matthews *et al.* 2000). The feedback relationships between stocks and flows described in Schandl and Schulz (2000) give some ideas as to future self-commitments of industrial societies.

Another important feature of the metabolic profile of industrial society is the overuse of the atmosphere as a sink. The main output category of disposals to domestic nature is CO_2, caused by fossil fuel use, animal husbandry and waste incineration. Industrial societies were environmentally successful in cleaning up water in the 1960s and in reducing local toxic air emissions by introducing end-of-pipe technologies. Currently, the problem of increasing waste amounts is often met by waste incineration, resulting in a problem shift from one gateway to another (for example, from the soil to the air). Since outputs like CO_2 cannot be reduced by waste treatment technologies, environmental problems shift from the local to the global level.

The remarkable similarities in industrial metabolism among many industrial economies encourage us to talk about a characteristic metabolic profile. Looking at the sheer level of average consumption it amounts to 18 tons per inhabitant and year (see Table 26.6). This should be further analyzed and discussed if there appears to be a different pattern within different groups of economies. On the one hand, there are Austria, Germany and the USA with a shared average of around 19 tons and, on the other hand, there are Japan and the Netherlands with an average of around 16 tons.

Table 26.6 A comparison of the material consumption in several industrial economies (tons per capita, 1991)

	Austria	Germany	Japan	Netherlands	USA	M*	UK
Biomass	4.8	2.6	1.5	4.3	3.0	2.9	1.5
Minerals	10.6	10.7	11.8	5.9	8.0	8.7	5.3
Fossil fuels	3.0	6.2	3.3	6.4	7.7	5.1	4.2
Products	0.1						0.2
Domestic material consumption	18.5	19.5	16.6	16.6	18.7	16.8	11.1
Population (millions)	7.8	80.0	124.0	15.0	252.8		57.8

Note: *Unweighted arithmetical mean for five countries.

Source: Adriaanse *et al.* (1997) for Germany, Japan, the Netherlands and USA; Schandl *et al.* (2000) for Austria; own calculations for the UK.

Data for the UK economy in this table seem to be outliers in this shared picture. This might be the case for various reasons; first, the accuracy of the available statistical data, especially true for construction materials, is still weak. Looking at the numbers, we consider biomass consumption and fossil fuels consumption data to be very reliable. Mineral consumption is quite low but, on the other hand, in a range with the Netherlands experience. Undoubtedly, the Netherlands is closest to the UK with regard to geomorphological features. Even if minerals consumption for the UK were to lie at around 6 to 7 tons, there seems to be slight evidence that the UK economy would follow a different pattern. One argument might be that the UK, as the entrepreneur of industrialization, has taken another historical path than that of later industrializers. As a result, the UK economy has already eliminated the most material-intensive heavy industry. The UK economy, for a rather long time, was a net importer of most resources. Finally, political decisions of the government in the 1980s may have hastened the advent of the 'new economy'.

CONCLUSION

On the basis of recent empirical case studies on economy-wide metabolism for several industrial economies, and as a result of international harmonization activities, we can discuss different metabolic profiles and can link them to socioeconomic development and environmental change. It seems that a specific mode of production and regime of accumulation contributes to a specific metabolic pattern, that is also subject to local resource availability. Future material flow analysis in the context of industrial ecology should focus on decomposition of trends and sectoral disaggregation. Also, research should link the local developments to global trends and should investigate global relations between specific economies. Furthermore, econometric analysis could be undertaken, since the available datasets are comparable now. All these future activities would strengthen the political approach and argumentation for an ecological modernization of the industrial system.

NOTES

1. Correlation analysis documents a stronger relation between population and materials input ($r = 0.961^{**}$) than between GDP and materials input ($r = 0.853^{**}$). This result indicates that developments in materials input might be explained by GDP or population. Further analysis should rely on regression analysis and t-test and should also test the variables for unit roots.

27. Industrial symbiosis: the legacy of Kalundborg

John R. Ehrenfeld and Marian R. Chertow

Much of industrial ecology is concerned with where resources come from – whether natural or man-made – and where they ultimately wind up. The focus can be on a single element such as lead or nitrogen, a single resource such as energy, or on multiple resources such as energy, water and materials. This focus is applied at different scales: from the facility level, to the inter-firm level, to a river or other regional site and, indeed, globally.

The branch of industrial ecology known as industrial symbiosis involves the physical exchange of materials, energy, water and by-products among several organizations. Thus, as indicated in Figure 27.1, it occurs at the inter-firm level. The keys to industrial symbiosis are collaboration and the synergistic possibilities offered by geographical proximity. As such, industrial symbiosis is not simply a passive examination or description of resource flows, but an active means of choosing the ones that are most useful in a localized economic system and arranging them accordingly. Ultimately, industrial symbiosis relies on a much different form of organization than is typical of conventional business arrangements. Therefore this chapter has two goals: (a) to discuss industrial symbiosis as a collective approach to competitive advantage through examination of an

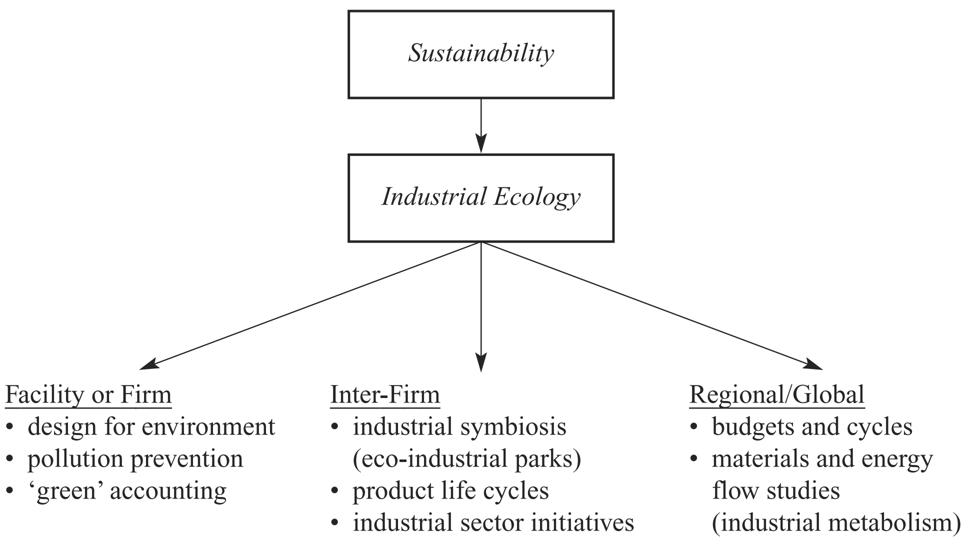


Figure 27.1 Industrial ecology operating at three levels

industrial district in Kalundborg, Denmark; and (b) to consider forms of industrial organization beneficial to advancing industrial symbiosis.

Symbiosis is a biological term referring to 'a close sustained living together of two species or kinds of organisms'. The term was used as early as 1873 by a German botanist, H.A. De Bary, to describe the intimate coupling of fungi and algae in lichens. While nature's living arrangements can be beneficial or harmful, the specific type of symbiosis known as mutualism refers to the situation in which at least two otherwise unrelated species exchange materials, energy or information in a mutually beneficial manner (Miller 1994). So, too, industrial symbiosis consists of place-based exchanges among different entities. It stresses collaboration, since, by working together, businesses strive for a collective benefit greater than the sum of individual benefits that could be achieved by acting alone. Such collaboration can also foster social values among the participants, which can extend to the surrounding neighborhoods. As described below, the symbioses need not occur within the strict boundaries of a 'park', despite the popular usage of the term 'eco-industrial park' to describe organizations engaging in exchanges.

The evolution of particular forms of industrial organization, that is, the way firms structure themselves to gain maximum competitive advantage, has long been a focus of economists. One of the dominant theories in this field is based on the notion that firms engaged in transactions (supply chains or extended product life cycles) will enter into whatever arrangements minimize the costs of these transactions (Williamson 1979). In the past, the environmental costs considered were relatively small and arrangements typically involved various forms of integration along the supply or value chain, such as traditional vertical integration in the iron and steel industry. More recently, transaction costs arising from proper environmental management have changed that calculus and new forms of organization are emerging to handle them. For example, the German packaging waste management system, Duales System Deutschland, is an independent company, funded by those firms that were made responsible under a German law for taking back packaging waste. The law created a new cost for these firms, in essence internalizing what had been an externality. The most economic organizational structure was deemed to be the consortium format that was adopted. The example of Kalundborg, Denmark, described below, is another window on the type of organizational structure that has evolved to re-use resources that would have been wasted and provides an outstanding example of the potential of industrial symbiosis.

KALUNDBORG AS A MODEL

The Kalundborg Complex: Historical Evolution

A highly evolved industrial symbiosis is located in the seaside industrial town of Kalundborg, Denmark (Gertler and Ehrenfeld 1996). Some 18 physical linkages comprise much of the tangible aspect of industrial symbiosis in Kalundborg (see Figure 27.2). The six key local players in the network that has developed are Asnaes Power Station, SK Power's 1350-megawatt power plant; a large oil refinery operated by Statoil A/S; Novo Nordisk Novozymes A/S, a Danish pharmaceutical and a Danish biotechnology company; Gyproc Nordic East, a plasterboard manufacturer; A-S Bioteknisk Jordrens, a

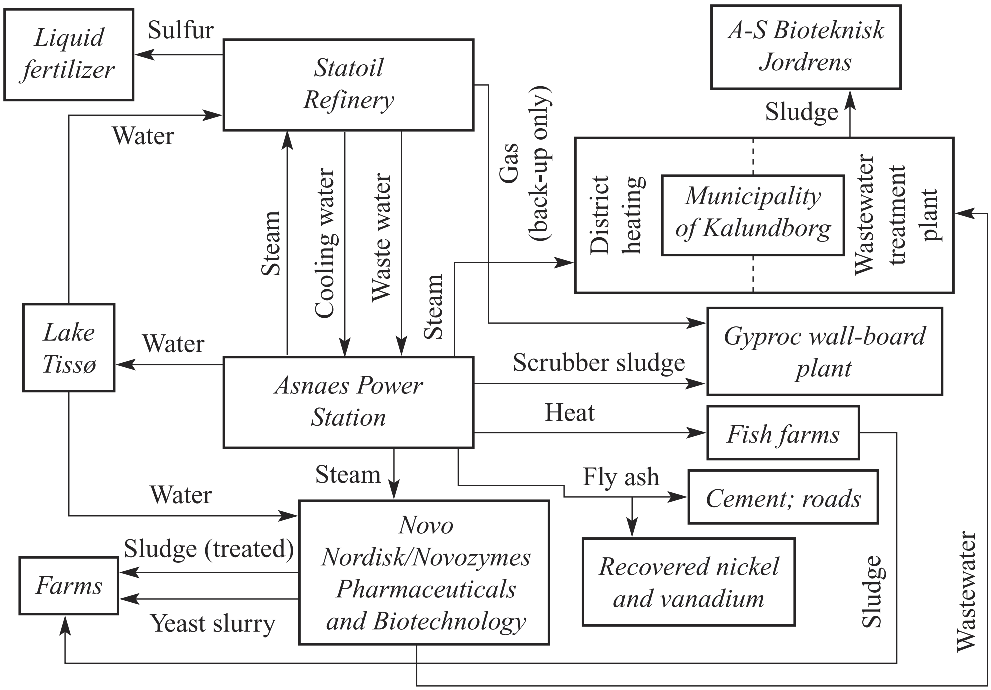


Figure 27.2 Industrial symbiosis at Kalundborg, Denmark

soil remediation company; and the municipality of Kalundborg. Several other users within the municipality trade and make use of waste streams and energy resources and turn by-products into raw materials. Firms outside the area also participate as recipients of by-product-to-raw-material exchanges. The symbioses evolved gradually (see Table 27.1) and without a grand design over the past 30 years, as the firms sought to make economic use of their by-products and to minimize the cost of compliance with new, ever-stricter environmental regulations.

At the heart of this system of arrangements is the Asnaes Power Station, the largest power plant in Denmark. Half of the Danish-owned power plant is fueled by coal and half by a new fuel called orimulsion, a bituminous product produced from Venezuelan tar sands. By exporting part of the formerly wasted energy, Asnaes has reduced the fraction of available energy directly discarded by about 80 per cent. Since 1981, the municipality of Kalundborg has eliminated the use of 3500 oil-fired residential furnaces by distributing heat from the power plant through a network of underground pipes. Homeowners pay for the piping, but receive cheap, reliable heat in return. The power plant also supplies cooling water that has been warmed 7–8 degrees in the process to supply an on-site fish farm producing about 200 tons of trout per year. Asnaes also delivers process steam to its neighbors, Novo Nordisk and Statoil. The Statoil refinery receives 15 per cent of its steam requirements while Novo Nordisk receives all of its steam requirements from Asnaes. The decision to rely completely on Asnaes for steam was made in 1982, when Novo Nordisk was faced with the need to upgrade and renovate its boilers. Buying steam from outside

Table 27.1 Chronology of Kalundborg development

1959	Asnaes Power Station commissioned
1961	Statoil refinery commissioned; water piped from Lake Tissø
1964	Original Novo Nordisk plant built
1972	Gyproc A/S built; excess gas piped from oil refinery
1973	Asnaes expands; draws water from Lake Tissø
1976	Novo Nordisk begins shipping sludge to farmers
1979	Asnaes begins to sell fly ash to cement producers
1981	Municipality of Kalundborg completes district heating distribution network, using steam from Asnaes Power Station
1982	Asnaes delivers steam to Statoil and Novo Nordisk
1987	Statoil pipes cooling water to Asnaes for use as raw boiler feed water
1989	Fish production begins at Asnaes site, using waste heat in salt cooling water
1990	Statoil sells molten sulfur to Kemira in Jutland (ends 1992)
1991	Statoil sends treated waste water to Asnaes for utility use
1992	Statoil sends desulfurized waste gas to Asnaes; begins to use by-product to produce liquid fertilizer
1993	Asnaes completes flue gas desulfurization project and supplies gypsum to Gyproc
1995	Asnaes constructs re-use basin to capture water flows for internal use and to reduce dependency on Lake Tissø
1997	Asnaes switches half its capacity from coal to orimulsion; begins to send out fly ash for vanadium/nickel recovery
1999	A/S Bioteknisk Jordrens uses sewage sludge from the municipality of Kalundborg as a bioremediation nutrient for contaminated soil

was seen as a cheaper alternative. The two mile-long steam pipeline built for the interchange paid for itself in two years. In addition, thermal pollution of the nearby fjord from the former Asnaes discharge has been reduced.

The power station also provides a gypsum-containing feedstock to Gyproc Nordic East, a neighboring wallboard maker owned by the British company BPB plc. In 1993, Asnaes completed the installation of a $115 million sulfur dioxide scrubber that produces calcium sulfate, or industrial gypsum, as a by-product. Conveniently, gypsum is the primary ingredient of wallboard and Asnaes' scrubber became the primary supplier of Gyproc's gypsum needs. In anticipation of a year 2000 tax on carbon dioxide, Asnaes sought additional CO_2 reduction and by 1998 had converted half of the plant from coal to orimulsion, described above. Achieving an 18 per cent CO_2 reduction actually increased the sulfur content of the scrubber sludge so that 170 000 tons of gypsum is now produced per year. Consequently, Asnaes now has the capability to meet all of Gyproc's gypsum requirement. Formerly, Gyproc obtained gypsum from a scrubber at a similar German power plant and also from Spanish open-pit mines. Both could provide back-up sources in the event that they are needed for smooth operations. Some 70 000 tons of fly ash, the remains of coal-burning power generation, is sold by Asnaes for road building and cement production.

The Norwegian-owned Statoil refinery, producing a range of petroleum products from light gas to heavy fuel oil, is located across the road from Asnaes, from which it draws 80 000 tons of steam. According to the environmental control officer (Ole Becher, personal communication 1998), the only by-product left from production of 4.8 million tons of crude oil per year is refinery gas which can be used internally or sold to Asnaes, once the sulfur is removed. In 1990, Statoil built a sour-gas desulfurization plant producing liquid sulfur that it shipped to a company for conversion to sulfuric acid. Today, about 20 000 tons of liquid fertilizer are manufactured from ammonia thiosulfate, which is a major by-product of Statoil's flue gas removal system, while the excess gas is burned at Asnaes. In 1972, Statoil began piping butane gas to Gyproc to fire wallboard drying ovens, all but eliminating the common practice of flaring waste gases. This system is now used as a back-up to public pipeline supply.

Groundwater scarcity in Kalundborg is generally claimed to be the motive force that brought many of the partners together (J. Christensen, personal communication 1998). In the early 1960s, need for surface water led to a Statoil project to bring supplies from Lake Tissø, some 50 kilometers from Kalundborg. Asnaes and Novo-Nordisk later joined the project as well. In addition, there are many other water and wastewater re-use schemes. Since 1987, Statoil has piped 700 000 cubic meters per year of cooling water to Asnaes, where it is purified and used as boiler feed water. Statoil has also made treated waste water available to Asnaes, which uses about 200 000 cubic meters a year for cleaning purposes. Statoil's investment in a biological treatment facility produces an effluent sufficiently clean for Asnaes' use. Symbiotic linkages have reduced total water consumption by participating companies by around 25 per cent and, at the power station, by 60 per cent.

A few kilometers from Asnaes and Statoil is Novo Nordisk, a world leader in the production of insulin and enzymes. The plant employs more than 1000 people. Novo Nordisk makes its product mix by fermentation, based on agricultural crops that are converted to valuable products by microorganisms. A nutrient-rich sludge remains after the products are harvested. Since 1976, Novo Nordisk has been distributing the process sludge to about a thousand nearby farms where it is spread on the land as fertilizer. After heat treatment to kill remaining microorganisms, the sludge is distributed throughout the countryside by a network of pipelines and tanker trucks. Novo Nordisk produces 3000 cubic meters of sludge per day, but can only store three day's worth. The sludge is given away instead of sold, reflecting the firm's concerns for disposal security. Three full-time employees coordinate its delivery. Distributing the sludge as fertilizer was the least-cost way to comply with regulations prohibiting Novo Nordisk from discharging the sludge directly into the sea. In addition, surplus yeast from Novo Nordisk's insulin production is sold as a high-value animal feed. Savings from more efficient utilization of resources and elimination of wastes are shown in Table 27.2.

Continuing Change at Kalundborg

Without careful analysis, it may seem that the effect of Kalundborg is to lock in old technologies in a situation of mutual dependence. The facts do not bear this out, but rather establish Kalundborg as a dynamic and adaptive system. Some trades have come and gone, such as Statoil's sulfuric acid production; some never got off the ground, such as greenhouses powered by Asnaes steam; and new ones are constantly being evaluated.

A new partner, A/S Bioteknisk Jordrens, joined the symbiosis in 1999. The company uses municipal sewage sludge as a nutrient in a bioremediation process to decompose pollutants in contaminated soils. This has allowed for beneficial re-use of another

Table 27.2 Waste and resource savings at Kalundborg

Annual resource savings through interchanges
Water savings
Statoil: 1.2 million cubic meters
Asnaes: total consumption reduction 60%
Fuel savings
Asnaes: 30000 tons of fossil fuel by using Statoil fuel gas
community heating via steam from Asnaes
Input chemicals/products
fertilizer equivalent to Novo Nordisk sludge (about 1300 tons nitrogen and 550 tons phosphorus)
97000 cubic meters of solid biomass (NovoGro 30)
280000 cubic meters of liquid biomass (NovoGro)
commercial fertilizers for 20000 hectares of farmland using Statoil sulfur
170000 tons of gypsum
recovered vanadium and nickel
Wastes avoided through interchanges
50000–70000 tons of fly ash from Asnaes
scrubber sludge from Asnaes
2800 tons of sulfur as hydrogen sulfide in flue gas from Statoil (air)
water treatment sludge from Novo Nordisk (landfill or sea)
380 tons of sulfur dioxide avoided by replacing coal and oil (air)
130000 tons of carbon dioxide avoided by replacing coal and oil (air)

material stream drawn from the city's waste water. Currently, some symbiosis partners are looking at other surface water sources to enable savings of groundwater and to offer alternatives at low flow times. The partners are also looking at water re-use broadly through the establishment of common water basins. The Asnaes Power Station recently added a 250000 cubic meter basin to improve water flow management.

The change at the power plant from coal to orimulsion has brought other trading opportunities by changing the effluent streams. The sulfur content in orimulsion is 2.5 per cent, versus 1 per cent for coal, so significantly more calcium sulfate is recovered from the flue gas desulfurization system and is available for raw material to make gypsum board. While the opportunity to close down older coal units has beneficial effects, orimulsion is not harmless. In fact, the fly ash has a significant heavy metal content with a concentration of 5 per cent nickel and 10–15 per cent vanadium. In addition to worker safety issues, two new exchanges have entered the symbiosis: the recovery and re-use of nickel and vanadium from the generator's ash stream.

A GENERAL FRAMEWORK FOR INDUSTRIAL SYMBIOSES

The Kalundborg complex is but one model of a symbiotic industrial organization. Chertow, following a detailed study of 18 potential eco-industrial parks examined at the Yale School of Forestry and Environmental Studies from 1997 to 1999, proposed a taxonomy of five different material exchange types (Chertow 1999a, 2000b). These are discussed here as Types 1–5 and are listed below:

- through waste exchanges (Type 1);
- within a facility, firm or organization (Type 2);
- among firms co-located in a defined eco-industrial park (Type 3);
- among local firms that are not co-located (Type 4);
- among firms organized 'virtually' across a broader region (Type 5).

Through Waste Exchanges (Type 1)

Many businesses recycle and donate or sell recovered materials through third party dealers to other organizations. Historically, scrap dealers have organized in this fashion, as have charities such as the Salvation Army. More recently, municipal recycling programs have become third parties for commercial and residential customers who supply recovered materials that are transported through the municipality to manufacturers such as glass plants and paper mills. This form of exchange is typically one-way and is generally focused at the end-of-life stage. Waste exchanges formalize trading opportunities by creating hard copy or on-line lists of materials one organization would like to dispose of and another organization might need. The scale of trades can be local, regional, national or global and can involve highly specialized chemicals or even lists of items needed by area charities. The exchanges accomplish various input–output savings on a trade-by-trade basis, rather than continuously. They feature exchange of materials rather than water or energy.

Within a Facility, Firm or Organization (Type 2)

Some kinds of material exchange can occur primarily inside the boundaries of one organization rather than with a collection of outside parties. Large organizations often behave as if they are separate entities and may approximate a multi-firm approach to industrial symbiosis. Significant gains can be made within one organization by considering the entire life cycle of products, processes and services, including upstream operations such as purchasing and product design.

Among Firms Co-located in a Defined 'Eco-Industrial Park' (Type 3)

In this approach, businesses and other organizations located in the equivalent of an industrial park can exchange energy, water and materials and can go further to share information and services such as obtaining permits, transport and marketing. Type 3 exchanges primarily occur within the defined area of the industrial park, but it is possible to involve other partners 'over the fence'. The areas can be new developments or retrofits of existing ones.

Among Local Firms that are Not Co-located (Type 4)

This type of exchange takes as a starting point what is already in place within an area, linking together existing businesses with the opportunity to fill in some new ones.

Kalundborg is an example of Type 4 exchange, in that the primary partners are not contiguous, but are within about a two-mile radius. Although this area was not planned as an industrial park, the proximity of the companies permitted them to take advantage of already generated material, water and energy streams.

Among Firms Organized 'Virtually' across a Broader Region (Type 5)

Given the high cost of moving and other critical variables that enter into decisions about corporate location, very few businesses will relocate solely to be part of an industrial symbiosis. In recognition of this, the model of Type 5 exchanges depends on virtual linkages rather than on co-location. While virtual eco-industrial parks are still place-based enterprises, Type 5 exchanges allow the benefits of industrial symbiosis to be expanded to encompass a regional economic community in which the potential for the identification of by-product exchanges is greatly increased owing simply to the number of firms that can be engaged. An additional attractive feature is the potential to include small outlying agricultural and other businesses, possibly by pipeline, as in Kalundborg, or by truck for those farther out. It could be argued that self-organized groups such as the network of scrap metal dealers, agglomerators and dismantlers who feed particular mills or subsystems such as auto-recycling could be considered as Type 5 virtual exchanges (Frosch *et al.* 1997).

The Underlying Dynamics of Symbiotic Evolutions

Material exchange types 3, 4 and 5 have many common characteristics with the more general notion of the manufacturing network form of industrial development presented by Piore and Sabel (1984) in their analysis of the success of the artisan-based economy in the Emilia-Romagna region of Italy. Active trade associations, shared services, such as purchasing and quality assurance, and close family and community ties are among the factors that contribute to the success of such industrial districts. Still, massive quantities of materials are routinely discarded as wastes by industrial systems throughout the world. This section discusses a set of factors that both promote and inhibit the development of symbioses and industrial ecosystems.

Many visitors come to Kalundborg looking for the master plan. Despite its impressive results, Kalundborg was not explicitly designed to demonstrate the benefits of industrial symbiosis. Each link in the system was negotiated, over a period of some 30 years (see Table 27.1), as an independent business deal and was established only if it was expected to be economically beneficial. Benefits are measured either as positive flows by marketing a by-product (or obtaining feedstocks at prices below those for virgin materials) or as savings relative to standard pollution control approaches. This is the strength of the Kalundborg approach: business leaders have done the 'right thing' for the environment in the pursuit of rational business interests. The evolutionary nature can be interpreted as pointing to a need to have both positive technical and economic factors appear simultaneously, a condition that may be difficult or impossible to realize in a forward-planning process.

Besides the basic chemical and other technical compatibility requirements of symbiotic partners, both need to recognize a net cost saving relative to their options. The floor for economic feasibility occurs when the difference in cost of the by-product feed relative to virgin or other alternatives is less per unit throughput than the cost of waste management to the producer. The user can offer more than enough to the producer to offset the costs of waste treatment or disposal. In practice the differential would also have to be large enough to account for transaction costs and risks to both parties. Typical transaction costs include regulatory, discovery, contracting and monitoring costs. Discovery costs, the costs required to learn of the existence of an opportunity for interchange, can be high, and may be the major impediment for material exchange of the types discussed above. Brokerages and markets serve to reduce these costs to the point that exchange is economically rational. Exchange of material recovered from municipal waste streams (for example, paper, metals, plastics) is now increasingly managed through commodity exchanges and electronic networks. The growth of Internet exchanges is likely to be a strong factor in reducing discovery and contracting costs.

The buyer of by-products in a symbiosis takes some risk by tying the firm to a single, outside supplier and to the vagaries of the supply continuity. However, the exchange of by-products and cascades of energy use is not inherently different from traditional supplier–customer relationships. Kalundborg's Gyproc, for example, maintains sources of gypsum other than the Asnaes Power Station so as not to disrupt operations during planned or unplanned power plant outages. Differential financial implications may be insignificant. Provisions for stand-by supplies will add cost. The seller also takes some risk owing to the possibility of upsets at the buyer's facility that could interrupt the outflow of the by-products. If this were to happen, the by-products would instantly become wastes to the seller and would need to be disposed of according to the relevant regulatory requirements.

Interestingly, recent experience in Kalundborg reinforces the point that the needs of the individual companies are of central concern. Over the last several years Kalundborg's Statoil Refinery has doubled its capacity based on North Sea claims, the Asnaes Power Station has switched from coal to orimulsion for half of its 1350MW of capacity to comply with mandated CO_2 reduction, and the pharmaceutical plant has also eliminated some product lines including penicillin and increased others. While each individual business change alters the make-up of the industrial symbiosis, they have not, collectively, diminished the spirit of it. In the case of the gypsum board plant, the changes have made the benefits stronger as more calcium sulfate is now produced.

Like most large industrial operators, the pharmaceutical plant management must meet annual continuous improvement goals in many areas, including established percentages for waste reduction (J. Christensen, personal communication 1998). Late in 2000, the enzyme business of Novo Nordisk was spun off as a separate company, Novozymes A/S. The stability of operations at Novo Nordisk and Statoil clearly depends on flows from the power plant. When asked what Statoil and Novo Nordisk would do if Asnaes were decommissioned, which is certainly plausible for a power plant, executives calmly said they would get together and build a steam plant. The cooperation that has developed has also led to non-material benefits since the companies have gotten together for personnel and planning sessions over the years.

Organizational Arrangements and Transaction Costs

Symbioses can follow, in theory, any of the common types of industrial organization described, for example, by Williamson (1979). Williamson suggests that organizational arrangements between firms are shaped by efforts to minimize transaction costs. Kalundborg is based on a complex of contracts and alliances that have arisen with little or no outside intervention from government or other sets of interests. Unlike a spot market such as is typical in handling metal scrap, this type of structure affords symbioses more certainty and continuity than exchanges in pure markets can offer. Vertical integration, the common ownership of one or more successive stages in the production process, would go even further and might arise if continuity in the movement of by-products became a critical factor.

Other forms of industrial organization more common outside the USA have some relevance to the emergence of symbiotic arrangements (Lenox 1995). The cross-ownership structure of the Japanese *keiretsu* is a highly elaborated form of integration in which the transaction costs and risks could be spread among potential participants in an exchange of by-products. Another possibility is central ownership as is found in Germany where banks may own substantial equity, and participate actively, in the management of a number of firms. Other financial institutions could play a similar role. Schwarz and Steininger (1995) point to an extensive recycling network among firms in the Styria region of Austria, acknowledging that the activity is largely unconscious and thus uncoordinated. To reduce the coordinative problems with eco-industrial parks discussed above, some form of common ownership or institutional management power vested in the developer of the park can improve the context for the emergence of symbiotic patterns. The Kalundborg partners now jointly support an information arm, The Symbiosis Institute (*www.symbiosis.dk*).

Impediments other than strictly economic ones exist as well, although Williamson might argue that all can be represented in terms of transaction costs. Symbiosis requires interchange of information about nearby industries and their inputs and outputs that is often difficult or costly to obtain. Kalundborg's small size of about 20000 residents and relative isolation have made for a tight-knit community in which employees and managers interact socially with their counterparts on a regular basis. This cultural feature leads to what a local leader calls 'a short mental distance between firms' (V. Christensen, personal communication 1994). Cultural pressures are also important. As in many Scandinavian settings, there is a backdrop of environmental awareness. In Kalundborg, no deliberate institutional mechanism was needed to promote conversations among the potential partners. Inter-firm trust is important in establishing alliances or contracts among participants (Gulati 1995). An atmosphere of trust in Kalundborg existed even in the absence of specific experience between firms.

Technical Factors

In general, symbiotic industrial facilities need to be in close proximity in order to avoid large transport costs and energy degradation during transit. High-value by-products such as pure sulfur from sour-gas treatment are exceptions. Contrary to the notion of pollution prevention and zero waste at a plant boundary, such as is, for example, the underlying policy goal of the US Pollution Prevention Act of 1990, symbiosis may work best when plants produce large quantities of waste. This situation seems to be contrary to the notion of eco-efficiency as applied to individual firms. It is not always best for either the bottom line or the environment to reduce a single plant's 'waste' to zero.

Industrial symbioses of Types 3 and 4 work best, if not require, at least one plant (anchor tenant) with large, continuous waste streams. Wastes that are largely organic in nature, like the effluent from fermentation of all sorts (pharmaceuticals or brewing), or raw agricultural or forestry wastes are attractive as it is the organic carbon that is useful as a feedstock. Supply security is important to the user of the by-product streams exactly as would be the reliability of an otherwise virgin feed supply. Use of organic streams from fermentation as feed or fertilizer requires assurance that toxic components or organisms are absent. Materials production, such as the manufacture of wallboard, is more technically challenging and requires much closer matching of compositions.

Early links at Kalundborg tended to involve the sale of waste products without significant pre-treatment. This pattern includes the initial sale of Statoil's flue gas, Asnaes' sale of fly ash, clinker, waste heat and process steam, as well as the use of cooling water to heat fish farm ponds. These arrangements simply involved rerouting of what was formerly waste, without significant alteration. The more recent links, however, have been created by and depend on the application of pollution control technologies. These links, which comprise just over half of the interconnections, do not just move process by-products around. The processes and disposal practices are controlled to make them more environmentally benign and, at the same time, to render them more attractive as feedstocks. The gypsum stream from Asnaes is the output of the flue gas control operating to remove sulfur dioxide which is present at low concentrations and in a chemical form that is not useful directly. The interposition of pollution control systems is important in an industrial ecosystem as these technologies serve to concentrate dilute by-products into economically and technically attractive forms. The symbiotic relationships that comprise these links would not be attractive in the absence of such pollution control measures.

Regulatory Context

The former manager of the Asnaes plant believes that existing economic incentives alone were generally sufficient for much of the Kalundborg symbiosis (V. Christensen, personal communication 1994). Further symbiotic arrangements yielding environmental benefits are potentially available, but cost more than conventional practices. Political impetus is necessary to go further, for example, requiring emission reductions or adjusting prices to make symbiosis economically attractive. Such external signals are not sufficient, however, since innovative and pioneering cooperation is required among companies for symbiosis to occur.

The Danish regulatory framework has encouraged the evolution of industrial symbiosis in Kalundborg. Compared to the USA, the Danish regulatory system is consultative, open and flexible. Rather than be reactive, firms are required to be proactive by submitting plans to the overseeing county government detailing their efforts to continually reduce their environmental impact. A dialogue then ensues in which the regulators and the firm establish goals. A more flexible, cooperative relationship is fostered between government and the regulated industries, and as a result firms tend to focus their energies on finding creative ways to become more environmentally benign instead of fighting with regulators. A key aspect of the flexibility is that regulatory requirements are mainly in the form of performance standards stating the degree of the desired decrease, instead of technology standards, as is common in the USA. Technology standards ensure that uniformly effective pollution control methods are adopted throughout a given industry. However, they tend to hinder technological or infrastructural innovation (Banks and Heaton 1995; Porter and van der Linde 1995; Sparrow 1994; Preston 1997). Many of the creative arrangements found in Kalundborg are only possible where firms have flexibility in the approaches employed to meet pollution reduction targets. In the USA, little discretion is left to firms. There are disadvantages to the Danish system, including potentially lower levels of technical compliance and high transaction costs incurred in extensive consultations about obtaining permits. Although US technology standards are inflexible, they ensure a certain minimum level of pollution control.

Regulatory requirements may preclude interchange or serve as very strong disincentives. In the USA, for example, the Resource Conservation and Recovery Act (RCRA) regulates the treatment and disposal of industrial waste, but inadvertently impedes one of its objectives – resource conservation and recovery. The statute is primarily concerned with averting risks stemming from the improper management of hazardous waste. RCRA regulations pursue this goal through a very extensive set of specific, inflexible and often confusing rules governing the treatment, storage and disposal of industrial by-products (Hill 1991). RCRA regulations set forth specific detailed procedural and technical requirements for the management of an exhaustive list of particular types of waste streams. With by-products being matched to a particular, mandatory protocol, there is little room for innovative schemes for their re-use as feedstocks elsewhere. This inflexibility is based in large part on a deep-seated fear of sham recycling, which is an undertaking where the generator of a waste product makes a show of re-using that by-product merely to escape treatment requirements (Comella 1993). Industrial symbiosis must be distinguished from such efforts if it is to develop within the current regulatory system.

FACILITATING THE EVOLUTION OF SYMBIOSES

Kalundborg illustrates that there is enormous potential for environmental improvement through industrial symbiosis. Positive applications include increasing energy efficiency through co-generation and by-product re-use, recycling gray (used) water to achieve overall reduction in drawdowns, recovering solvents and re-using many, diverse residue streams that need not be rejected as wastes. Other non-material-based linkages, such as jointly planning transport networks and sharing office, information, or security services, also have potential for environmental improvement. Given these advantages, one might ask why more companies are not engaged in these types of projects.

Basic Economics

Some regard Kalundborg as a singular historical phenomenon, the unique conditions of which are unlikely to be reproduced. First of all, there are the usual business reasons why such projects might not be attractive, based on barriers any venture faces: risk, finance, mobility of capital or the availability of higher pay-back options elsewhere. Reliable research is clearly needed on the basic economics of symbiosis. If energy or water or waste disposal are but a small percentage of operating costs, these reasons alone will not cause the formation of eco-industrial parks. There must be sufficient quantities of materials to make exchanges worth while. Neither can there be fixed heuristics about when symbiosis makes sense, since, for example, fresh water could be scarce at one site and abundant at another. As with all environmental projects, particulars are site-specific and the role of regulation is ubiquitous, in both promoting and obstructing progress, and must be carefully considered in these non-traditional development projects.

The Actors

Although private actors need not be the initiators, they clearly must be committed to the implementation of industrial symbiosis since, in most instances, the industrial symbiosis flows either belong to private companies or will be shared with them in the case of municipal waste water linkages. This is where the perseverance of 'business as usual' presents a significant barrier since many of the costs and benefits of industrial symbiosis fall first to private actors and then to the community at large. Whether the private actors can appropriate sufficient benefit from environmental gains is a challenge to industrial symbiosis. As a practical matter, all significant development projects take a long time and a lot of effort. This is compounded with eco-industrial park projects by the need for multi-party planning and coordination and the attendant transaction costs. Indeed, even explaining industrial symbiosis – the educational component – is arduous because industrial symbiosis is not business as usual, and requires a significant change to traditional individualist mental models.

Pollution Prevention v. Symbiotic Flows

Industrial symbiosis raises the question of whether the desire to re-use waste streams comes at the expense of adhering to pollution prevention principles calling for the elimination of waste at the front end of the process. The same arguments could be applied to oversupply of water or energy, thus discouraging conservation. At the first level of analysis, it is reasonable to assume that companies will do what is in their economic interest. If, through incremental improvements or through broader-scale process redesign, a company can eliminate waste in a cost-effective manner, then it will. In this sense, pollution prevention comes first. It is plausible, however, that the opportunity for symbiosis might make the proposed process improvement fall lower in priority in a company's capital outlay scheme, in which case the company's own economic decision making might favor the symbiosis over pollution prevention.

Some question whether eco-industrial parks favor older, dying industries and keep them going rather than fostering a new generation of clean technology. Overall, industrial symbiosis could discourage companies from updating their systems, plant and equipment, substituting, instead, the veil of interdependence. Recent experience at Kalundborg, however, suggests that symbioses can remain robust in the light of changes even promoting shared innovation among the participants.

An industrial ecology perspective offers another cut at the issues raised above. Is waste a waste or an unused raw material? Industrial ecology, by demanding a systems approach, gives due consideration to each step and stage of process development to optimize material and energy flows. In some, or even most, cases, reduction of a waste stream may be called for; in others, using a particular stream to feed another business may be optimal, depending on related logistics, economic considerations and the state of technology. The analytic question is straightforward: which configuration leads to the lowest level of environmental damage at a given level of economic output?

Architect/designer William McDonough and chemist Michael Braumgart also question current practice by asking whether eco-efficiency is a viable strategy: successive 10 per cent reductions, following Zeno's Paradox, never get you to zero, and certainly not to the goal the authors establish of regeneration rather than depletion. McDonough and Braumgart (1998) make the point that nature itself does not seek efficiency as a goal; for example, plants may spawn thousands of seeds while only a few germinate. Thus they refer to nature's bounty not as eco-efficiency but as eco-effectiveness: 'highly industrious, astonishingly productive and creative'. By analogy, it is reasonable to conclude that industrial symbiosis may not appear to be an eco-efficient solution in every case, but it may often be an eco-effective one.

THE IMPORTANCE OF EVOLUTIONARY APPROACHES

Currently, interest in industrial symbiosis is running high, from clusters of brewers and cement manufacturers in Japan (Kimura and Taniguchi 1999) to government planning in the Philippines (Bateman 1999), to sustained Canadian emphasis (Environment Canada 1997) and global curiosity. To date, however, few eco-industrial parks have broken ground. The most significant conclusion to be drawn reinforces what was experienced in Kalundborg: cooperation develops over time. Therefore evolutionary approaches are key to moving industrial symbiosis forward.

One approach, known as *green twinning* or *by-product synergy*, consists of a single material or energy exchange. The exchange stands on its own environmentally and economically but, by example, could lead to other types of exchanges. Typical instances would be co-generation of steam and electricity, use of recirculated water or conversion of ash into a building material. Each has the potential to be the initial stage of broader industrial symbiosis. In Texas, Chaparral Steel and its related company, Texas Industries, developed a newly patented process to add slag from the steel plant to the raw material cement mix of Texas Industries (Forward and Mangan 1999). As a result, cement production increased by 10 per cent and energy consumption dropped by more than 10 per cent. The value of the slag increased by 20 times over the previous market price offered by road contractors and landfill costs dropped significantly. Moreover, the twinning has led to additional by-product re-use including baghouse dust drawn from air filtering equipment and automobile shredder residue.

In Kalundborg, companies did not become partners to work on industrial symbiosis, but came to their partnership through *organizational relationships* begun to solve a common problem: the need to find a surface water source. From this relationship, other symbiotic ideas emerged. A by-product synergy project in Tampico, Mexico, organized through the Business Council for Sustainable Development of the Gulf of Mexico, relied on an existing industry association in the Tampico-Ciudad Madero-Altamira region for a demonstration project there. The final report notes that the project was able to take advantage of the association's structure and relationships (Business Council for Sustainable Development – Gulf of Mexico 1999).

A third evolutionary approach, borrowing elements from the other two, has been dubbed the *anchor tenant model*. Just as shopping malls are built around several large department stores that anchor the commercial development within, one or two large industries can provide the same critical mass for an eco-industrial park. AES power plants are anchors for developing projects in Guayama, Puerto Rico and Londonderry, New Hampshire (Chertow 2000b). An existing nuclear plant anchors the Bruce Energy Center in Tiverton, Ontario, which incorporates a hydroponic greenhouse, a food processor and a manufacturer of commercial alcohols to take advantage of waste heat and steam generation from the plant (Peck and Ierfino 1998). This concept is very important, given the restructuring in the electricity industry, because every new power plant could become the anchor tenant of a surrounding eco-industrial park (Chertow 1999b). While the barriers to successful, conscious industrial symbiosis are many, the legacy of Kalundborg has inspired one of the great metaphors of industrial ecology: the industrial ecosystem. Key to the implementation of industrial symbiosis is sufficient economic incentive, technological cooperation and great human perseverance.

PART V

Industrial Ecology at the Sectoral/Materials Level

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28. Material flows due to mining and urbanization

Ian Douglas and Nigel Lawson

Mining and urbanization involve the greatest transformations of the landscape through human activity. Mining may leave huge pits and waste heaps, while urban areas contain large stocks of materials brought in from other places. Minerals extraction is broadly divided into three basic methods: open-pit or surface, underground and solution mining. Open workings are the dominant form of extraction of the main commodities mined or quarried: coal and aggregates. Surface, or open-pit, mining requires rock, soil and vegetation removal to reach mineral deposits. The waste rock, or overburden, is piled near the mine. The workings have large energy requirements and produce emissions to the atmosphere and discharges to nearby water bodies. For any particular mine, these hidden flows are often greater in magnitude than the mass of mineral or ore extracted for processing.

Urban use of materials involves two broad strands of inputs, stocks and outputs. The buildings and infrastructure of the city can be described as the 'urban fabric' (Douglas 1983) while the materials (largely food) consumed by the people and all other organisms within the city can be seen as passing through the urban biosphere. The biospheric use of materials has a rapid turnover, expressed by the high proportion of food waste and packaging in the domestic waste stream. The biospheric consumption is largely biomass-derived food and clothing, water and energy mainly using fossil fuels, but an increasing amount of hydrocarbon synthesized materials are used in packaging and other short-life materials. The inputs to the urban fabric include wood from biomass, but are mainly from mining and quarrying, and thus in national assessments of domestic and imported mineral products. The urban fabric has a slower turnover as buildings last for decades, if not centuries and, in rare cases, millennia. The renewal of the urban fabric produces construction and demolition waste (C&D waste) much of which is used to level the original site for new construction. Over the centuries, this leveling gradually raises the level of city streets and building ground floors over the residues of past structures. The residues thus become part of the 'urban deposit' (Wilburn and Goonan 1998). The dumps of wastes from biospheric consumption, from the industrial transformation of materials and from disposal of C&D waste are also part of this urban deposit. The urban fabric, and all the materials housed and stored within it, and the underlying and surrounding urban deposit make up the urban materials stock. The outputs from the city are all the transformed, manufactured and processed materials and goods as well as the gaseous emissions and liquid discharges and solid materials released to the surrounding environment.

Despite having relatively static total populations, the industrialized economies are increasing their use of materials, particularly construction materials, as households become smaller but more numerous and individuals acquire more possessions. In the rapidly industrializing countries, urban and industrial building and the construction of roads and other infrastructure are proceeding apace. For example, in China, the production of aggregates and associated hidden flows more than doubled in seven years, rising from 2313 million metric tonnes (MMT) in 1989 to 5403MMT in 1996 (Chen and Qiao 2000). However, per capita flows remain highest in the wealthier countries, the flows associated with aggregates being approximately 7.8 metric tonnes (t) per person per year in the USA, approximately 4.2t per person per year in China and only 0.39t per person in India.

This rapid transfer of materials from the natural environment to urban and industrial areas has a twofold impact: a removal of material from the earth's surface (a change in geomorphology) and the accumulation of a stock of concrete and other materials elsewhere in cities and industrial zones (a change in urban morphology). Currently, in many places, waste flows also lead to morphological change as landfills occupy old quarries or parts of river floodplains, or develop new hills as a land raise (a landfill in which the deposited material rises above the general level of the surrounding area). Thus industrial ecology transforms natural landscapes and in so doing has to be considered as a geological and geomorphological agent.

The role of human activity in earth surface processes and geological transformations has long been acknowledged. Sherlock's excellent (1922) studies provide many illustrations of the quantities of material involved in mining, construction and urban processes. In the concern over making land use and urban life more sustainable in the 1990s, much greater attention than ever before has been paid to the ecological footprints of cities (Rees 1994) and the ecological 'rucksack' of mining (Bringezu and Schütz 1996). Already analyses of materials fluxes have been produced for China, the USA, Japan, Germany, the Netherlands and Italy (Chen and Qiao 2000; Adriaanse *et al.* 1997; de Marco *et al.* 1999). Girardet has estimated the ecological footprint of Greater London as 125 times the area it occupies (Sustainable London Trust 1996). Earth scientists concerned about the human dimensions of geological processes have established a program (ESPROMUD) to demonstrate the environmental footprint of cities and extractive industries and to define guidelines to reduce it which specifically aims to assess the effects of extractive and urbanization activities on geomorphic processes (Cendrero and Douglas 1996; Douglas and Lawson 1997a). Accounting for the flows of materials in the urban process is a key element of this program.

The key quantities to be addressed in establishing the materials flows due to mining and urbanization are the annual masses of rock and earth surface materials extracted, including overburden and mineral processing residues, the volumes of waste created by human activities and the excavation and earth moving involved in major construction projects such as tunneling and road building. In addition, account should be taken of pollutant releases to the atmosphere, water and the soil, as well as of the chemicals, fuels and materials in machinery used in the mining and mineral-processing activities.

METHODOLOGICAL DEBATE

National mining and quarrying statistics record the net amounts of material produced for each commodity extracted, the run of mine production. However, these figures do not account for all the overburden removed before extraction can start and the waste created during mineral processing, the hidden flows associated with mining and quarrying (Adriaanse *et al.* 1997; Douglas and Lawson 1997a; Ayres and Ayres 1999b; de Marco *et* *al.* 1999). While it can be argued that, during open-pit mining, especially coal mining, much of the overburden and locally stored waste is eventually replaced in the hole from which it came, even that local temporary shift of materials involves a large energy consumption and a change in the nature of the ground surface. In addition, there is always a risk that some of the stockpiled overburden will be eroded or lost to the people-modified, natural drainage system and that the final land restoration will result in a somewhat different landscape from that which existed before mining. The quantities of material shifted by mining and quarrying will vary with the mineral being extracted, the technology used to extract it, the geological situation of the deposit, the age and life of the mine, and the precise management policies of the mine operator. Nevertheless, this 'ecological rucksack' of mining and extractive industries has to be estimated.

Researchers, including Sherlock (1922), Hooke (1994), Warhurst (1994), Weizsäcker *et al.* (1997) and Adriaanse *et al.* (1997), have all tried to gain an idea of these 'hidden' flows using estimated multipliers of the mineral production to obtain a figure for the total amount of materials shifted by mining and quarrying. For each mineral product, a given additional quantity of earth surface materials is removed as overburden or as waste from ore treatment (Figure 28.1). The ratio of this additional quantity to the amount of mineral produced can be used as a multiplier to obtain the total volume of material moved. To obtain the multipliers used in this chapter, case study examples of mine overburden quantities cited in the literature were examined for more general applicability and approximate multipliers were established for as many mineral commodities as possible. These approximations were then extensively revised through consultation with a wide range of experts involved with the mineral industries (Douglas and Lawson 1998). This provides a list of multipliers based on commodities than can be applied to world figures. Their application to individual countries may be less safe, particularly if a single mine, or group of mines, in an unusual geologic formation is responsible for virtually all the production of a particular mineral in that country.

INFORMAL AND UNRECORDED MINING ACTIVITY

National mineral production statistics vary in accuracy, and there is probably a certain amount of unrecorded, and often illegal, removal of gravel and rock in every country. Some countries have a large amount of small-scale artisanal mining, some of which may be unregulated and unrecorded. Gold Fields Mineral Services of London's estimate of 175t of gold being produced in 1995 by informal miners worldwide may be an underestimate (M.M. Veiga, personal communication 6 July 1998). Three million carats of diamonds may be mined informally each year in Africa (Holloway 1997). Because of low recovery rates when informal artisanal gold and gem miners open up sites with very low yields, these operators probably shift more than twice the mass of material per net tonne of pure mineral extracted than do their formal mining counterparts. L.K. Jéjé (personal communication 1998) suggests a ratio as high as 3.7 million to 1 for artisanal gold mining in the Amazon, as compared with a ratio of about 1 million to 1 for large-scale industrialized gold mining. These operations alone could easily result in the movement of over 400MMT of material per annum, equivalent to nearly 1 per cent of the global mass moved during mineral extraction.

In many parts of the developing world, building materials are worked as small enterprises. People seek the clay, sand, gravel and stone that they need in the nearest most

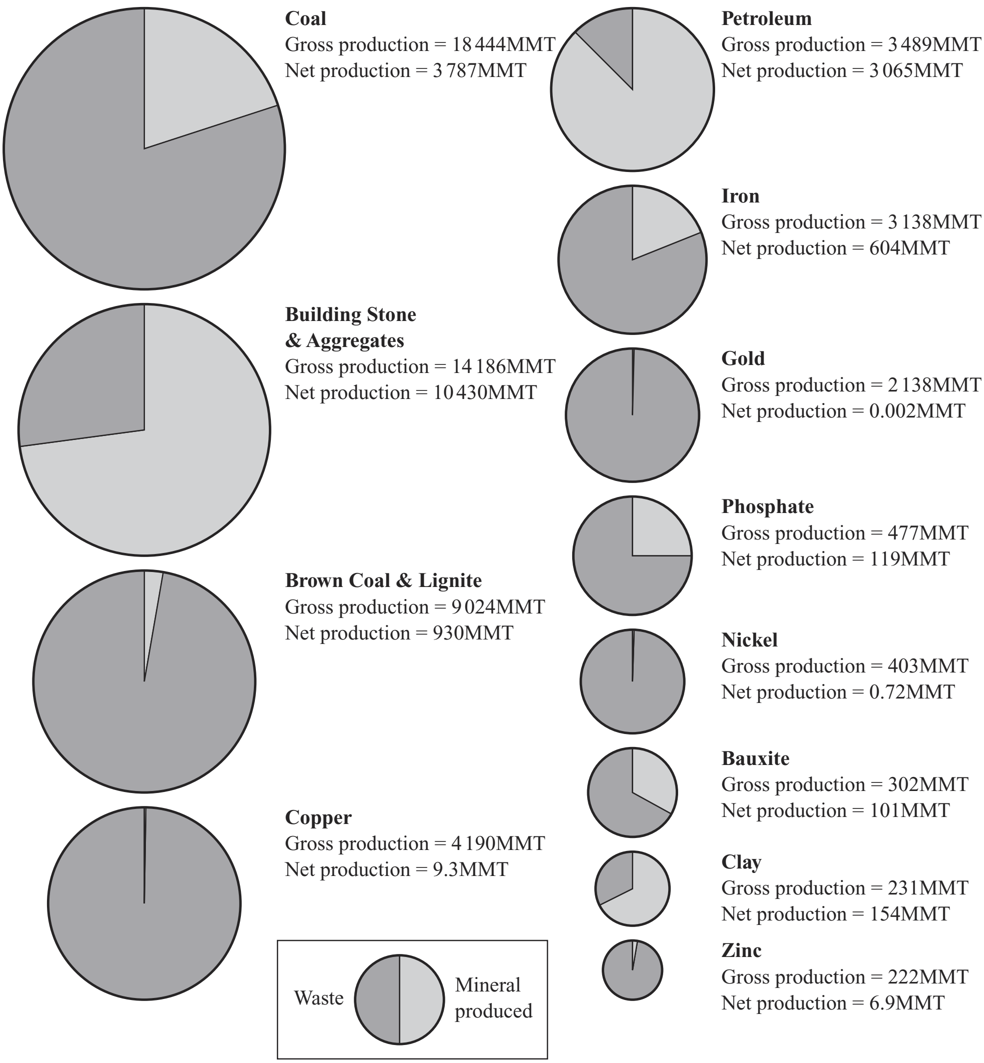


Figure 28.1 World mineral production and total 'hidden flows' for the 12 commodities producing the largest total materials flows at the global level

convenient place and it is inconceivable that these activities are accurately recorded. In Vietnam, for example, at headlands along the coast, individual entrepreneurs quarry granite, while around towns and cities small brick clay pits leave a series of derelict hollows and degraded soils. In inland China, villagers operate small crushers making aggregates from rock quarried from tiny rock outcrops near their fields. Many hundreds of thousands of tonnes of material are worked in this way and in clay pits (Edmonds 1994) and inefficient unauthorized mines (Qu and Li 1994), thereby degrading potential agricultural land. Road construction in remote areas, such as logging roads in Borneo, uses up to 370t of gravel per kilometer (km), and may involve the unrecorded quarrying of over 4000t of rock per year in each major Borneo logging concession (assuming some 10km of road construction or repair each year).

The global amount of material moved by these informal, and largely unrecorded, mining activities is difficult to assess. Assuming that such mining involves 250t per person per year (R. Nötstaller, personal communication 25 July 1998) and noting that in India alone about 200 000 informal miners (Chakravorty 1991) may be shifting about 50MMT of material each year, about 2.5 per cent of all mining-related materials displacement in India (Lawson and Douglas 1998). While these activities result in substantial amounts of earth movement, they probably add no more than 1 to 5 per cent to the global mass of materials moved during the extraction of minerals (Douglas and Lawson 2000a).

URBAN MATERIALS FLOWS

Since 1965, many attempts have been made to establish urban materials use (Wolman 1965; Aston *et al.* 1972; Newcombe 1977; Newcombe *et al.* 1978; Douglas 1983). Nevertheless, even the amount of mineral matter entering the urban fabric is difficult to establish, as facing and flooring materials, such as marbles, granites and tiles, are often imported. At the present state of knowledge, for any given country, it is usual to assume that all the bricks, aggregates and cement used come from national mineral production.

Another type of materials flow is involved with road construction and infrastructure development. Masses of earth surface material are removed in foundation excavation, trenching, making cuttings, tunnels and embankments. Statistics of these quantities are not easily obtained, but accounts of major projects in the engineering literature usually give a global figure of the amount of earth moving involved. For example, some 20MMT of material was excavated and moved during the construction of the Channel Tunnel (CIRIA 1997; Varley and Shuttleworth 1995) while the building of a 3.1km-long second runway at Manchester Airport involved the importation of over 1MMT of concrete aggregates and the excavation of 2.8 million cubic meters (Mm^3) of earth (A. Jack, earthworks agent, Amex–Tarmac Joint Venture, personal communication 24 November 1997).

The construction of road surfaces will be accounted for in assessments of mining and quarrying and in quantities of construction and demolition waste and road planings (scrapings from used road surfaces) recycled into aggregates for road making. However, the building of roads and highways also requires moving substantial amounts of earth – cut and fill. Each individual section of road will of course vary in accordance with the natural topography. In the UK, for example, recent motorway projects required the movement of approximately 500 000 tons per km of earth and rock and, on average, earth and overburden has to be removed to a depth of 0.75m to make way for the foundations for new urban construction and the provision of minor service roads (Douglas and Lawson 2001).

WASTE FLOWS

Establishing the outputs of urban waste products is a key issue in quantifying urban materials flows. Many estimates of per capita waste production are found in the literature, but it is unclear whether such figures apply to total waste volumes or to domestic, or domestic and municipal, waste only. In total, domestic and municipal waste accounts for only 1.5 per cent of the estimated 10 billion MMT of waste produced annually in the USA. Of this waste, 75 per cent is related to the environmental 'rucksack' and other hidden flows of mining and oil and gas production, 9.5 per cent to industry, 13 per cent to agriculture and 1 per cent to sewage sludge (Miller 2000).

Generally, waste statistics are not totally reliable. For example, in the UK, despite increased regulation of waste flows, data on the volumes involved are still inaccurate and those available only apply to licensed waste disposal sites and do not take account of exempted waste disposal schemes and the, probably not inconsiderable, amount of uncontrolled, unofficial illegal dumping, known locally as 'fly-tipping', throughout the country. Controlled waste data from UK government sources and the Environment Agency of England, Wales and Northern Ireland will almost certainly underestimate the total waste volume. However, in terms of urban materials budgets, this underestimate may be counterbalanced by some possible double accounting in terms of construction waste movements, with a proportion of such waste being taken to landfill and this becoming part of both the materials movement during construction and the flow of waste materials.

Throughout the world per capita municipal and industrial solid waste generation continues to increase. However, the per capita figures vary widely from city to city, depending largely on average wealth. In Abidjian Africa in 1994, 200kg of waste per head were produced, while at the same time, Washington, DC produced 1246kg per capita (World Resources Institute 1996). In many cities, particularly in the poorer parts of the world, much solid waste is not collected properly, and accumulates in piles in streets or between dwellings and factories. Few effective inventories of the total waste produced can be made in such cities. However, sometimes artisanal or informal waste collection, or 'rag picking', can recycle large volumes of usable materials that would be dumped in more affluent cities.

Reclamation of paper and cardboard by small entrepreneurs is highly organized in cities as diverse as Nairobi, Calcutta, Cairo and Beijing. Recycling is increasing in many countries, but much more remains to be done (see Chapter 44). In Denmark, taxes on many types of solid wastes have increased recycling to over 61 per cent of household waste generation. In the UK, landfill and aggregate taxes have been introduced to encourage recycling of potential wastes but only some 8 per cent of household waste was being recycled or composted in 1998 (UK Department of the Environment, Transport and the Regions 1999). Some 53.6MMT of C&D waste are produced annually, but only some 5MMT per year of such materials are re-used for civil engineering and building construction. The potential, however, is not always matched by practical feasibility, within the context of both current standards/specifications and geographical location, where low price and the cost of transport to areas of substantial demand discourages use of these waste materials. The government aims to increase the use of secondary aggregates to 40MMT per year by 2001 and to 55MMT per year by 2006 (UK Department of the Environment 1994a). However, variations in composition are a major problem. While UK demolition waste comprises approximately 41 per cent by weight concrete, 24 per cent masonry, 17 per cent paper, cardboard and plastic and ceramics, metals and other materials, 15 per cent asphalt and 3 per cent wood-based products (Hobbs and Collins 1997; CIRIA 1997), construction waste may contain 45 per cent soil and other active surface materials (UK Department of the Environment 1994b).

Estimates of total amounts of demolition and construction waste produced in England and Wales vary (Howard Humphries and Partners 1994; Symonds Group Ltd. 1999). Of the approximately 53.6MMT per year:

- 27.4MMT per year are deposited in landfill. Perhaps 20 per cent of this material is employed in engineering works on site (haul and access roads, construction of cells, cover and so on);
- 21.2MMT per year are exempt from licensed disposal and are used in unprocessed form or coarsely crushed for use in demolition/construction sites and for sale/disposal off site for land modeling during the construction of projects such as golf courses and equestrian centers;
- 5MMT per year only comprise material which is either crushed to produce a graded product or is directly recovered.

In addition, material scraped from the surface of bituminous road pavements produces around 7.5MMT of material per annum and much of this finds its way to secondary uses as capping layers, public footpaths or haul/access road construction (UK Department of the Environment 1994b).

The destinations of urban waste are changing. Much legislation encourages recyling and discourages landfill. In Europe, the traditional dumping of sewage sludge in the sea ceased at the end of 1998 as a result of a European Community Directive. Some sludge is now converted into energy by incineration. Other sludge is spread on agricultural land while some ends up in landfill (Priestley 1998). Such changes in waste flow paths alter the destinations of substances contained in sludge, such as heavy metals like cadmium which may eventually find their way back into the food chain (see Chapter 33).

MATERIALS BUDGETS AT THE URBAN AND REGIONAL LEVELS

Materials flows from the agricultural biomass support life in urban areas. Timber is also consumed in large quantities, but the rate of flow through the system is slower than that of food for wood used in construction and furniture, but almost the same rate as food for newsprint and packaging materials. Estimates of these national biomass flows can be obtained from FAO, *Agricultural and Forestry Statistics* (annual), while studies of national metabolism combine domestic production figures with imports and exports of biomass products (Schandl and Schultz 2000). Urban metabolism data incorporating these materials flows were generated by the pioneering work on Sydney and Hong Kong (Aston *et al.* 1972; Newcombe 1977) and in later studies such as those on Vienna (Daxbeck *et al.* 1997) and Taipei (Huang 1998).

Regional materials flow accounts assist in examining regional sustainability, especially when combined with the regional energy consumption. In Upper Austria, such an analysis of the construction sector showed that, when urban areas are expanding, the construction materials flow into the urban fabric is likely to be three times the amount of material entering the C&D waste stream or being recycled (13 to 19t per capita per year input, but only 4.1 to 6.3t per capita per year C&D waste) (Glenck and Lahner 1997). River basin-based analyses of natural and people-driven materials flows offer an alternative to regional accounts. Investigations in the moist temperate environment of northern Spain and in the humid tropical environment of Puerto Rico indicate the scale of materials shifted by extractive industries against the work of natural fluvial processes. The Spanish Besaya Basin has 13.9MMT per year of material excavated for mining and another 4.2MMT for urbanization and infrastructure development, but the river only removes 0.07MMT of sediment per year to the sea. For the Rio Loiza in Puerto Rico, 1.24MMT of sand and gravel are dug out of the river channel for urban construction each year, while the sediment yield to the river is 2.80MMT (Douglas and Lawson 1997a).

CONCLUSION

Materials flows for mining and urbanization make up most of the total mass domestic extraction in countries like the UK (510MMT out of 587MMT in 1997 in the UK, according to Schandl and Schultz 2000). Hidden flows, mainly the removal of overburden, accentuate this dominance. Globally, mining and quarrying produces 19 735MMT of minerals but involves the shifting of a total mass of 57 549MMT, indicating that the hidden flows are about three times greater than the actual production (Douglas and Lawson 1997a).

By country per capita materials shifts vary (Table 28.1) from 4.87t per capita per year in India to 53.65t per capita per year in the USA. Such comparisons indicate how much greater the reduction in per capita materials consumption will have to be in some countries if the goal of greater equity in sustainable development is to be achieved.

Table 28.1 Totals of materials moved by the main types of extractive industry, infrastructure development and waste creation activities in selected countries (flows in MMT per year)

	USA	Germany	Japan	UK	Netherlands	China	India
Fossil fuels production	1684	365	13	54	68	1570	242
Fossil fuels hidden flows	5846	2333	3	305	1	8400	1014
Industrial minerals production	105	53	194	13	7	171	56
Industrial minerals hidden flows	312	35	21	26	1	2234	201
Construction materials production	1730	749	1103	313	59	4912	312
Construction materials hidden flows	159	164	0	122	15	491	63
Infrastructure cut and fill	2956	300	1105	180	27	22208	2452
Dredging	516			21	24		
Waste generation MSW	180	28	50	20	7	435	38
Waste generation industrial/C&D	921	232	395	101	22	580	181
Total materials moved	14111	4259	2884	1155	231	41091	4559
Population (millions, 1995)	263	82	125	56	16	1221	936
Per capita annual materials movement (t)	53.65	51.93	23.07	20.63	15.00	33.65	4.87

Sources: USA, Germany, Japan and Netherlands from Adriaanse *et al.* (1997); China materials from Chen and Qiao (2000); population from World Resources Institute (1996); waste from Douglas and Lawson (1998). Other data from authors' calculations.

APPENDIX: GLOBAL MATERIALS FLOWS ASSOCIATED WITH MINERALS EXTRACTION, INCLUDING HIDDEN FLOWS

The total mineral production recorded in published data for 1995 was 19.7 billion tons, but the actual total amount of material mined or quarried from the earth's surface was of the order of 57.5 billion tons as some 37.8 billion tons of waste, overburden or spoil were also moved (Table 28A.1). The multiplier for coal plays a major role in the magnitude of the global figure of total material extracted. Although most overburden removed during open cast operations will be put back, we assume no overburden replacement in this assessment of earth surface change even though good site restoration programs can, in time and in certain locations, successfully return the land to up to 80 per cent of original productivity.

There are few figures covering the developing world. The only available global estimates of aggregates and building materials production have been made by correlating US production figures with GDP (Evans 1993; Hooke 1994). AS GDP includes exports, it is going to be a variable predictor. Alternatively, an extrapolation could be based on population or energy consumption. As development involves building roads, housing and industrial facilities which use both aggregates and energy, data on energy consumption may be a good predictor of aggregate use. Absence of efficiency considerations in energy figures may be compensated for by the additional labor use. In the available national UN aggregate and building stone production data the production figures correlate better with energy consumption than with GDP or population:

aggregates, energy	80 per cent of results are within	9 per cent +/− of the median
aggregates, GDP	80 per cent of results are within	40 per cent +/− of the median
aggregates, population	80 per cent of results are within	100 per cent +/− of the median

Table 28A.1 Global mineral production and associated earth materials movement, 1995

Commodity	Production net weight (MMT)	% World	Multiplier	Production gross weight (MMT)	% World	Remarks
Coal, hard	3787	19.19	4.87	18444	32.05	
Building stones: granite, porphyry, sandstone including marble and travertines	10430	52.85	1.36	14186	24.65	Net production figures: by correlation with energy consumption
Aggregates: limestone flux and calcareous stone, gravel and crushed stone, sand, silica and quartz[1]						
Coal, brown + lignite	930	4.71	9.9	9204	15.99	
Copper ores – Cu content	9.3	0.05	450	4190	7.28	
Petroleum, crude[2]	3065	15.53	1.02	3489	6.06[1]	Gross weight includes 375 million tons oil shale and oil sand production wastes
Iron ores – Fe content	604	3.06	5.2	3138	5.45	
Gold ores – Au content	0.002	0.00001	950000	2138	3.71	Net production: Nötstaller (1997)
Phosphates, natural	119	0.60	4	477	0.83	
Nickel ores – Ni content	0.72	0.003	560	403	0.70	
Bauxite, crude ore	101	0.51	3	302	0.52	
Clay	154	0.78	1.5	231	0.40	
Zinc ores – Zn content	6.9	0.035	32	222	0.38	
Salt	17	0.84	1	166	0.29	
Gypsum, crude	99	0.50	1.2	119	0.21	
Kaolin	28	0.14	4	114	0.20	
Diamonds, industrial and gem (221738 thousand carats)	0.00005	0.00	2380000	109	0.19	
Lead ores – Pb content	2.7	0.014	32	88	0.15	
Ilmenite – concentrates	3.4	0.017	25	85	0.15	
Manganese ores – Mn content	11	0.056	6	67	0.12	
Peat, for agriculture and fuel	42	0.21	1.25	52	0.09	
Iron pyrites, unroasted	7	0.04	5	36	0.06	
Gasoline, natural	36	0.18	1	35	0.06	
Uranium ores – U content	32.97	0.0002	900	33	0.06	
Fuller's earth	8.0	0.04	4	32	0.06	

Bentonite	7.32	0.04	4	29	0.05	
Chalk	23	0.11	1.2	27	0.05	
Potash salts – K_{20} content	22	0.11	1	22	0.04	
Tin ores – Sn content	0.18	0.001	100	18	0.03	
Andalusite	1.24	0.006	9	11	0.02	
Magnesite	9.5	0.05	1.2	11	0.02	
Abrasives, natural	8.7	0.04	1.2	10	0.02	
Fluorspar	4.0	0.02	2	8.0	0.01	
Barytes	4.2	0.02	2	8.5	0.01	
Slate	5.4	0.03	1.5	8.1	0.01	
Talc	6.5	0.03	1.2	7.8	0.01	
Chromium ores – Cr content	3.3	0.017	2	6.8	0.01	
Tantalum and niobium concentrates	0.05	0.0002	100	4.6	0.008	
Asbestos	2.5	0.01	1.5	3.7	0.007	
Borate minerals	3.4	0.02	1	3.4	0.006	
Natural gas	2.6	0.01	1	2.6	0.005	BGS World Mineral Statistics, conversion rate: $0.022m^3 = 20$ gr
Tungsten ores – W content	0.024	0.0001	100	2.4	0.004	
Antimony ores – Sb content	0.15	0.0007	9	1.3	0.002	
Graphite, natural	0.65	0.003	2	1.3	0.002	
Arsenic trioxide	0.045	0.0002	20	0.91	0.002	
Cobalt ores – Co content	15.20	0.00008	20	0.30	0.0005	
Vanadium ores – V content	0.032	0.0002		0.00	0.00	By-product
Zirconium concentrates	0.94	0.005	100	0.00	0.00	By-product
Silver ores – Ag content	13.26	0.00007		0.00	0.00	By-product
Sulfur	18	0.09		0.00	0.00	By-product
Total	**19735**	**100**		**57549**	**100**	

Notes:

[1] Aggregates and building stone production accounts for over 52 per cent of the net weight of all mined or quarried material. Published UN data only cover 22 countries who produce about 50 per cent the aggregates and building stones used.

[2] Oil won from oil-bearing shales and sands is included in the net figure for production of petroleum, crude. However, oil from these sands and shale deposits requires additional materials movement of 22 times the net quantity of oil produced. Oil products produced from oil-bearing shales and sands by the major producing countries amounted to 17 857 thousand tons in 1994 (United States Bureau of Mines 1994).

Sources: Adapted from Douglas and Lawson (1997b); production figures, net weight (unless otherwise stated) from Department of Economic and Social Affairs, Statistical Division of the United Nations (1995).

Table 28A.2 Estimated total annual production and stockpiles of waste materials in the UK, by sector

Sector	Annual production MMT	%	Remarks	Stockpile	Stockpile and disposal to landfill; region where available
Agriculture	80	19	Wet weight, housed livestock		
Colliery spoil	19	5	Excludes overburden	3600	Coal mining areas
Slate	7	2		450	N. Wales, Lake District, S.W. England
China Clay	24	6		600	Cornwall and Devon
Quarrying	32				
Sewage sludge	36	9	Wet weight, av. 4% solids		
Dredged spoils	33	8	All UK waters, external and internal		
Municipal waste	29	7	27MMT household		25 MMT to landfill annually
Commercial	15	12			12 MMT to landfill annually
Demolition and construction	66	17			30 MMT to landfill annually
Blast furnace and steel slag	6			12	N.Yorks, Humberside, Wales
Power station ash	13	12			Power stations
Other	55			150	Spent oil shale in the Lothian region of Scotland and 41 MMT of industrial and other waste to landfill annually
Total	415	100			

Sources: Annual production, UK Department of the Environment, Transport and Regions (1997); annual production and disposal to landfill, adjusted from the figures for England and Wales to allow for Scotland on the basis of population, from A. Bell, regional waste strategy manager, The Environment Agency (personal communication 17 December 1997); stockpiles, UK Department of the Environment (1995).

Table 28A.3 Summary of controlled waste in England and Wales, production and disposal

Waste type	Total production MMT/year	Landfilled %	Incinerated %	Other disposal* %	Recycled %
Commercial and industrial	82.4	60.6	2.0	17.5	19.9
Demolition and construction	53.5	51.2		39.6	9.2
Municipal and household	25.8	88.6	5.0		6.4
Sewage sludge (dry solids)	1.0	10.5	8.1	29.8	51.6
Total	162.7	61.6	1.9	22.1	14.4

Note: *Predominantly in house disposal, for example fly ash, and waste disposal which is exempt from licensing such as land spreading paper pulp and food waste, material which can benefit agricultural land and construction wastes used in land modeling schemes. Charges and the imposition of a landfill tax have increased abuse of exemption schemes.

Sources: UK Department of the Environment, Transport and the Regions (1997); UK Department of the Environment (1995), (1994b); UK Secretary of State for the Environment and the Secretary of State for Wales (1995); Water Services Association (1996); Bell (1997); personal communication, A. Bell 17 December 1997.

Table 28A.4 Sludge production and disposal methods in a selection of countries

Country	1985 Population (millions)	1984 Dry solids (1000t/y)	Disposal method (% of total sludge production) Agricultural use	Landfill or stockpile	Incineration	Ocean dumping
UK	56	1 018	45.0	21.0	3.0	30.0*
W. Germany	59	2 180	32.0	59.0	9.0	0.0
Japan	120	1 133	8.4	35.2	54.6	1.6
Australia	17	300	9.0	76.0	2.0	13.0
USA	235		24.7	48.4	21.4	5.5

Note: *Ocean dumping by EC countries ceased in 1998.

Source: After Priestley (1998).

Table 28A.5 Earth removal during some major tunneling and civil engineering projects in the UK

Project	Earth excavated	Remarks	Source
Channel Tunnel	20Mt	Excavated chalk marl transformed into 40ha of remedial parkland at the foot of Shakespeare Cliff	CIRIA (1997); Varley and Shuttleworth (1995)
London Underground Jubilee Line Extension	3.5Mt	Most spoil disposal to landfill, including substantial quantities of contaminated soils. Small quantities to Thames reclamation sites	Personal communications, R. Humphries (13/8/1997), Public relations manager, Jubilee Line Extension Project; H. Shaw (9/12/1997), ex-logistics manager, Jubilee Line Extension Project
Conway Tunnel (A55)	$4Mm^3$	Up to $2.7Mm^3$ re-used as fill. Construction of the reclamation area to take unsuitable material required importation of 130 000Mm^3 stone rip-rap	Davies *et al*. (1990)
Dinorwic Pumped Storage Power Station	$5.25Mm^3$	Lower Lake excavation · $4Mm^3$. Tunnels and caverns $1.25Mm^3$	Personal communication, E. Snowden (8/1/1997), Kier Construction Limited, Civil Engineering Division
Sellafield (nuclear waste storage)	$1.1Mm^3$	Plans to mid-21st century entail the excavation of about $7Mm^3$ of material	Ball *et al*. (1996); Personal communications, B. Breen (22/10/1997), Nirex Limited; B. Paul (31/7/1997), manager, British Nuclear Fuels Limited, Drigg
Manchester Airport Runway 2	$2.80Mm^3$	Cut and balance Construction of the 3100m runway and taxiways also requires the importation of 1Mt concrete aggregates	Personal communication, A. Jack (24/11/1997) earthworks agent Amex–Tarmac Joint Venture
Thames Water Ring Main	1Mt	Predominantly dense London clay, used for sealing gravel pits and capping landfill sites	Personal communication, P. Claye (29/7/1997), Thames Water Utilities, Engineering

29. Long-term world metal use: application of industrial ecology in a system dynamics model

Detlef P. van Vuuren, Bart J. Strengers and Bert J. M. de Vries*

Over the last century, the exploitation of material resources has grown enormously. Currently, western economies use about 20 to 40 metric tons of raw materials per person per year (Adriaanse *et al.* 1997). While high material consumption rates certainly have contributed to the high living standards in large parts of the world, their enormous throughput has also raised questions with regard to the sustainability of current use. Especially during the energy crises in the 1970s, several authors have pointed out the risks of depleting reserves of high-grade resources; predictions were made that the world would run out of some raw materials in 50 years (for example, Meadows *et al.* 1972). At the moment, attention seems to have shifted to the question of whether ore grade depletion might aggravate the environmental problems associated with metal production (Tilton 1996). Clearly, exploitation of raw materials requires a sizeable amount of global capital and energy inputs and causes different sorts of environmental problems in mining, transport and upgrading. In addition, virtually all materials ultimately return to the environment, creating fluxes of substances that are potentially harmful to the environment. Industrial ecology intends to introduce integrated responses to this type of problem. System dynamics models form one of the tools that contribute to this. In this chapter, we will focus on a system dynamics model for an important type of material use, that is, metals.

Earlier, production and consumption of metals have been analyzed using material flow analyses (for example, Moll 1989; Jolly 1993; Annema and Ros 1994; van der Voet 1996). Attempts have also been made to analyze resource use in the broader context of economic growth and technological development, sometimes explicitly related to industrial ecology (for example, Suzuki and Shoji 1977; Chapman and Roberts 1983; Gordon *et al.* 1987; De Vries 1989a, 1989b; Duchin and Lange 1994; Ayres and Ayres 1996; Weston and Ruth 1997). Building on such analyses, we have developed a system dynamics model which simulates the long-term structural dynamics of metal resource exploitation and which in principle can be linked with larger integrated assessment models. The model can be of assistance in exploring the issue of sustainability of metal resource use, especially in relation to population and economic growth, on the one hand, and consequences for

* This contribution is based on van Vuuren, Strengers and de Vries (1999), 'Long-term perspectives on world metal use – a system-dynamics model' Resources Policy 25, pp. 239–55, with permission from Elsevier Science.

energy and capital requirements and waste flows, on the other. The main focus of the contribution here is to describe the model and apply it to some important trends in metal resource use between 1900 and 1990. We use the model to briefly explore possible futures that have been described in more detail elsewhere (van Vuuren, Strengers and de Vries 1999).

USE OF METALS BETWEEN 1900 AND 1990 AND THE METALS MODEL

Generally, three types of metals are distinguished on the basis of their resource availability: abundant metals (such as iron and aluminum), medium scarce metals (such as copper and lead) and scarce metals (such as gold and platinum). Within the metals model, we have concentrated on the first two groups, in particular iron and a virtual 'metal' called MedAlloy aggregated from all medium scarce metals (Cu, Pb, Zn, Sn and Ni); see de Vries (1989b). In the case of abundant metals, it was decided to focus solely on iron, since the flows of other abundant metals are small compared to those of iron. A similar model has been constructed for each 'metal', with the two models operating independently.

The model's general structure is a simple flow diagram (Figure 29.1). Geological resources are converted into reserves by exploration activities. Geological resources refer to the total amount of a metal available in the earth's crust and can, in the time span of the model, be considered as almost infinite. Depletion is modeled in terms of a decreasing ore grade of reserves, as discussed further in this chapter. Ore extracted from the reserves is used to produce refined metals or metal compounds (primary production), which are subsequently used to produce final consumption goods. These products remain in use for some time, during or after which the metals they contain either slowly dissipate or are dumped in places where they could constitute a secondary resource as long as they are not dissipated into the environment. Alternatively, materials can be recycled directly after their lifetime (secondary production). Consumption in the model is defined as primary production plus secondary production.

The stocks and flows are governed by various information flows; Figure 29.2 presents the most important ones. In terms of its dynamics, the model combines several theories into one integrated model: in particular, analysis of metal demand on the basis of intensity of use analysis, the interplay of technological development and ore grade decline, and competition between primary and secondary production. In the rest of this section, these elements will be briefly described. A more detailed description can be found elsewhere (van Vuuren 1995). The most crucial model equations have been displayed in Appendix A.

METAL DEMAND

Over the last 100 years the global consumption of both iron and MedAlloy has sharply increased, with an average annual growth rate of 3.2 per cent and 2.9 per cent, respectively, as shown in Table 29A.1. Obviously, the building of cities, heavy industry and the like have made a significant contribution to the growing use of metals. In the last two

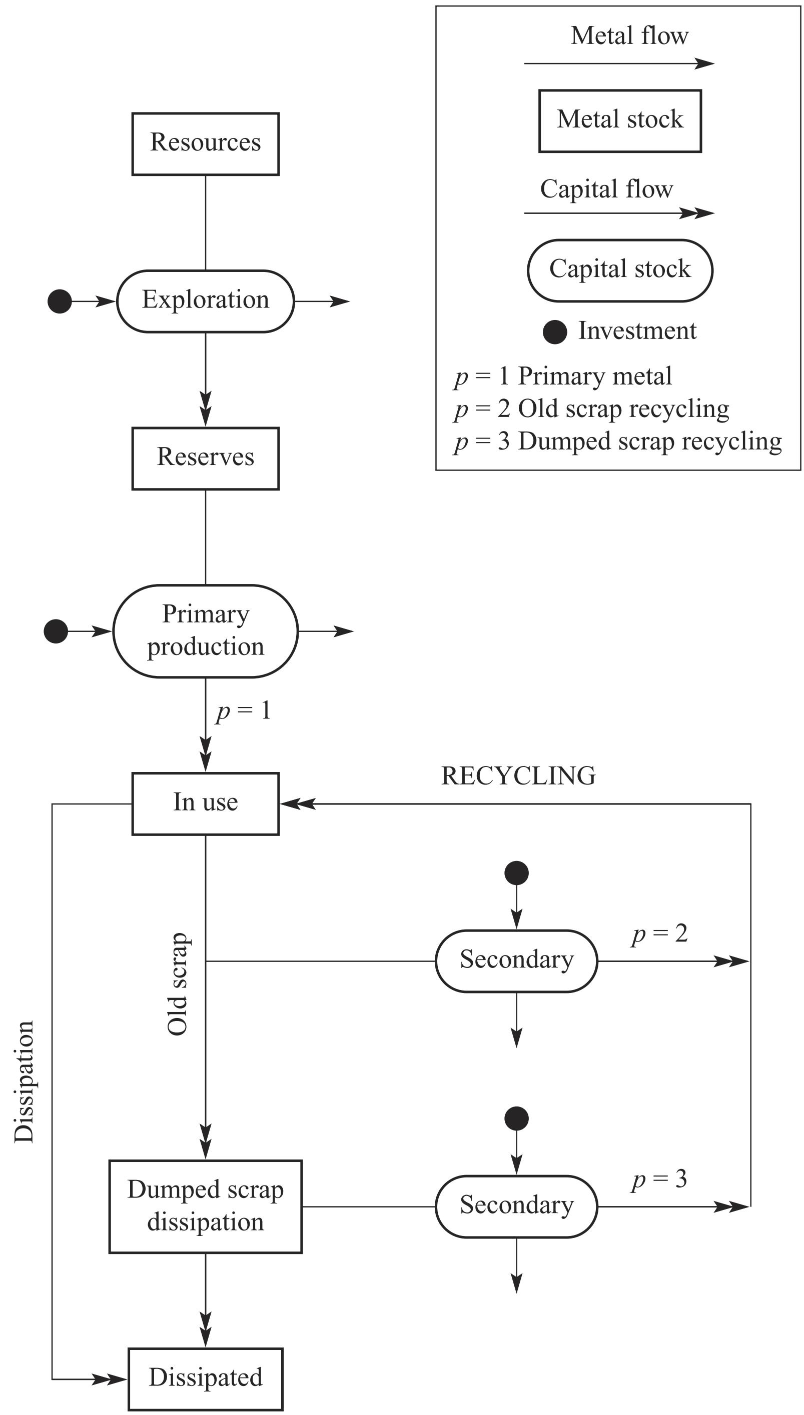


Figure 29.1 Stocks and flows in the metal model for iron/steel and MedAlloy

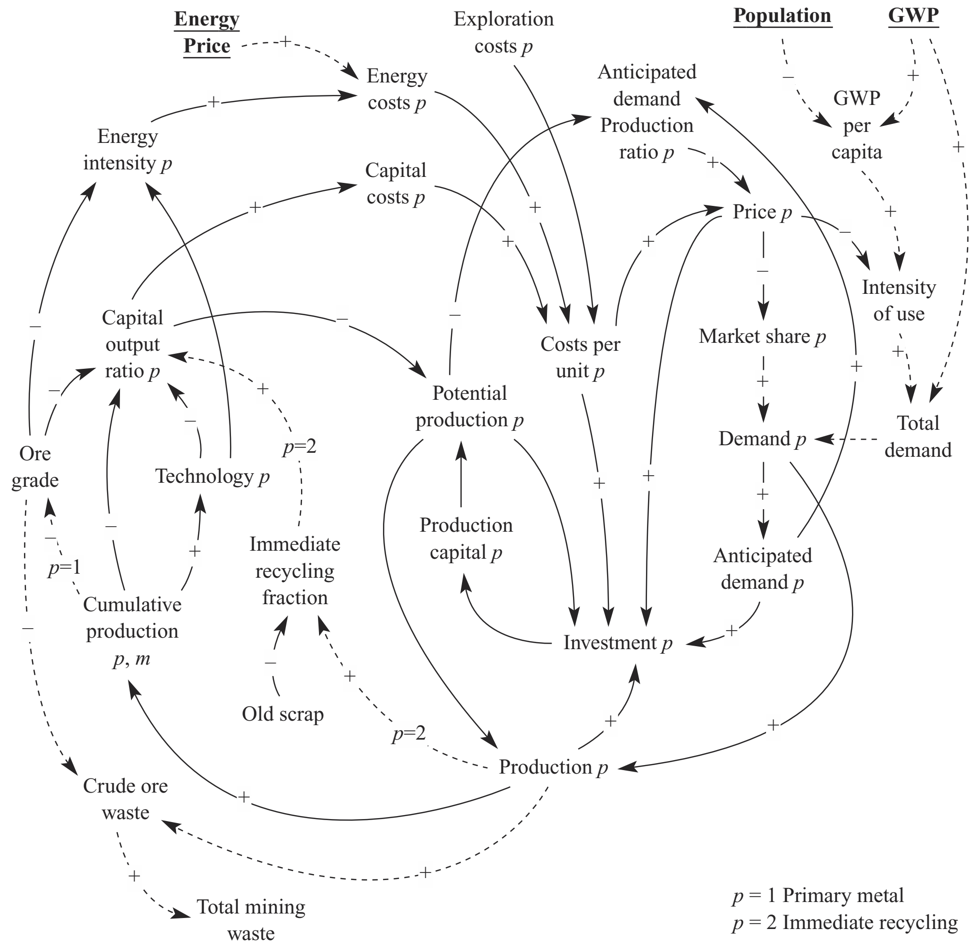


Figure 29.2 Model relationships within the metal model

decades, demand for most metals in many industrialized countries has grown only very slowly or even leveled off. In contrast, demand for iron and other metals has been growing sharply in many developing countries. Recently, Asia experienced an average annual growth in iron demand of about 8 per cent per year. In 1992, the per capita consumption of steel in developing countries was on average 51kg and in industrialized countries over 300kg, indicating the large potential for growth in the former.

Empirical research in resource economics has found that metal use intensity (defined as metal consumption per unit GDP) can often be described as a function of per capita income. This function varies among countries and materials, but its general shape follows an inverse U-shaped curve (Malenbaum 1978; Altenpohl 1980; Tilton 1990b; Roberts 1996). Similar inverted U-shaped functions are sometimes also found for environmental pressure (referred to as environmental Kuznetz curves) see, for example, de Bruyn *et al.* (1998). The inverted U shape can be explained in terms of superposition of three different trends (see Figure 29.3).

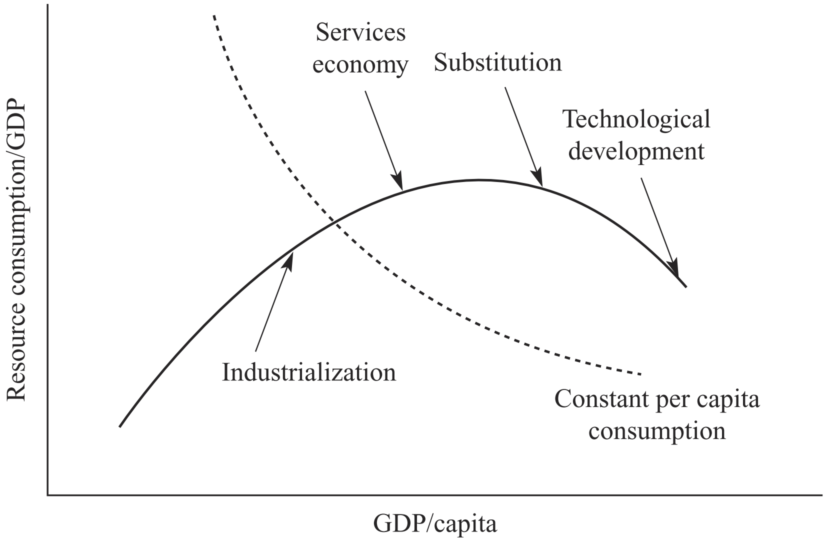


Figure 29.3 Intensity of use hypothesis

1. Intensity of use (IU): the changes in metal requirements in different phases of the *economic transition* from agriculture (low IU) to manufacturing and construction (high IU) and then to services (low IU) (see Tilton 1986). For a wide range of countries, the size of the different sectors has been found to correlate (at least partly) with per capita income (for example, Maddison 1989). The shift to a higher share of manufacturing and construction requires large (material) investments in building industrial infrastructure. In addition, it should be noted that the imports and exports of metals embodied in final products (such as cars) are not recorded in trade statistics. Therefore reallocation of final product industries to countries with lower labor costs and related exports to high-income countries will tend to decrease the IU in the latter and increase the IU in the former regions.
2. The changes in metal requirements as a result of *substitution*: the demand cycle of a material generally follows a pattern, in which the first stage of rapid growth after introduction is followed by a stabilization phase and a final phase, in which the markets for the material become saturated. At the same time, cheaper or better materials (for example, plastics) might penetrate the market and replace the original material. The reversal of growth can be so complete that even per capita or absolute consumption levels may begin to decline.
3. The changes in metal requirements as a result of *technological development*, which lead to more efficient raw material use in the production of final goods or satisfaction of consumer functions (for example, less metals use per car, as described in Forester (1988); Benardini and Galli (1993)).

The three processes involved can explain to a great extent the observed dematerialization in more advanced economies and will be a key factor in understanding future trends. The advantage of the IU hypothesis is its simplicity; its most serious shortcoming is that neither new technology nor material substitution depends primarily on changes per capita income. Obviously, the IU hypothesis is also unable to deal with unexpected breaks from the past, such as an energy crisis.

Figure 29.4 shows the IU for the consumption of steel and MedAlloy for 13 world regions. The data covers the 1970–90 period except for the USA, for which the period 1900–1990 is used (Schmitz 1979; van Vuuren 1995; Klein-Goldewijk and Battjes 1997). Real GDP per capita is used on the basis of constant purchasing-power-parity corrected dollars (ppp$) of 1990, taken from Summers and Heston (1991). Use of ppp$ instead of exchange rate-based dollars allows for better cross-regional comparison. It should be noted that both metal demand data and GDP data are somewhat uncertain and often difficult to compare. Therefore comparison of different regions and trends should be interpreted carefully. The dashed curves in Figure 29.4 are isolines representing a constant per capita consumption of metals.

The history of steel consumption in the USA is generally regarded as an excellent example of the IU hypothesis. Since about 1920, consumption per unit of GNP has decreased to 40 per cent of the peak level. Interestingly, the historical trend of the USA and the position and trends of the other regions follow an inverted U-shaped curve reasonably well. Four regions diverge sharply from this general pattern: the Former Soviet Union and Eastern Europe (high metal intensities during the centrally planned period, corresponding to the importance of heavy industry, followed by a collapse of metal use during the economic restructuring process), Middle East (relatively high national income due to oil exports) and Latin America (monetary data suffers from high inflation rates; foreign debt and high capital charges might prevent development of basic industries). The great differences between the regions, their different directions of development and the position of the global average in Figure 29.4 (indicated under 'World') all illustrate that it is impossible to determine long-term global trends directly from average global income and metal consumption. Economic growth in current low-income (and industrializing) regions could well cause the global average curve to diverge from its present dematerializing trend as shown in the scenario studies in this chapter.

All individual metals included in MedAlloy and many other metals follow trends similar to those in Figure 29.5. The most important exception is aluminum, for which the IU in high-income countries has not started to decline. Aluminum is in fact replacing other metals for various kinds of functional use (Moll 1989).

It should be noted that in Figure 29.5, technology transfer from high-income regions to low-income regions could be a fourth dynamic factor (next to economic transition and substitution, and technical development) determining the shape of the IU curve. Benardini and Galli (1993) suggested that the maximum intensity of the curve declines if reached later in time by a given economy.

IU models have also been constructed for specific country at the level of sectors. However, owing to the limited data availability at the global level, we have focused on metal use in the economy as a whole, which has been shown to give good results (Roberts 1996). On the basis of Figure 29.4, the model uses two single IU functions for iron and MedAlloy in combination with regional income and population development to determine regional metal demand. The IU function in the model can be changed with regard to its shape and the level at which per capita metal consumption is finally saturated (see also Appendix for the most important model equations). In addition, the

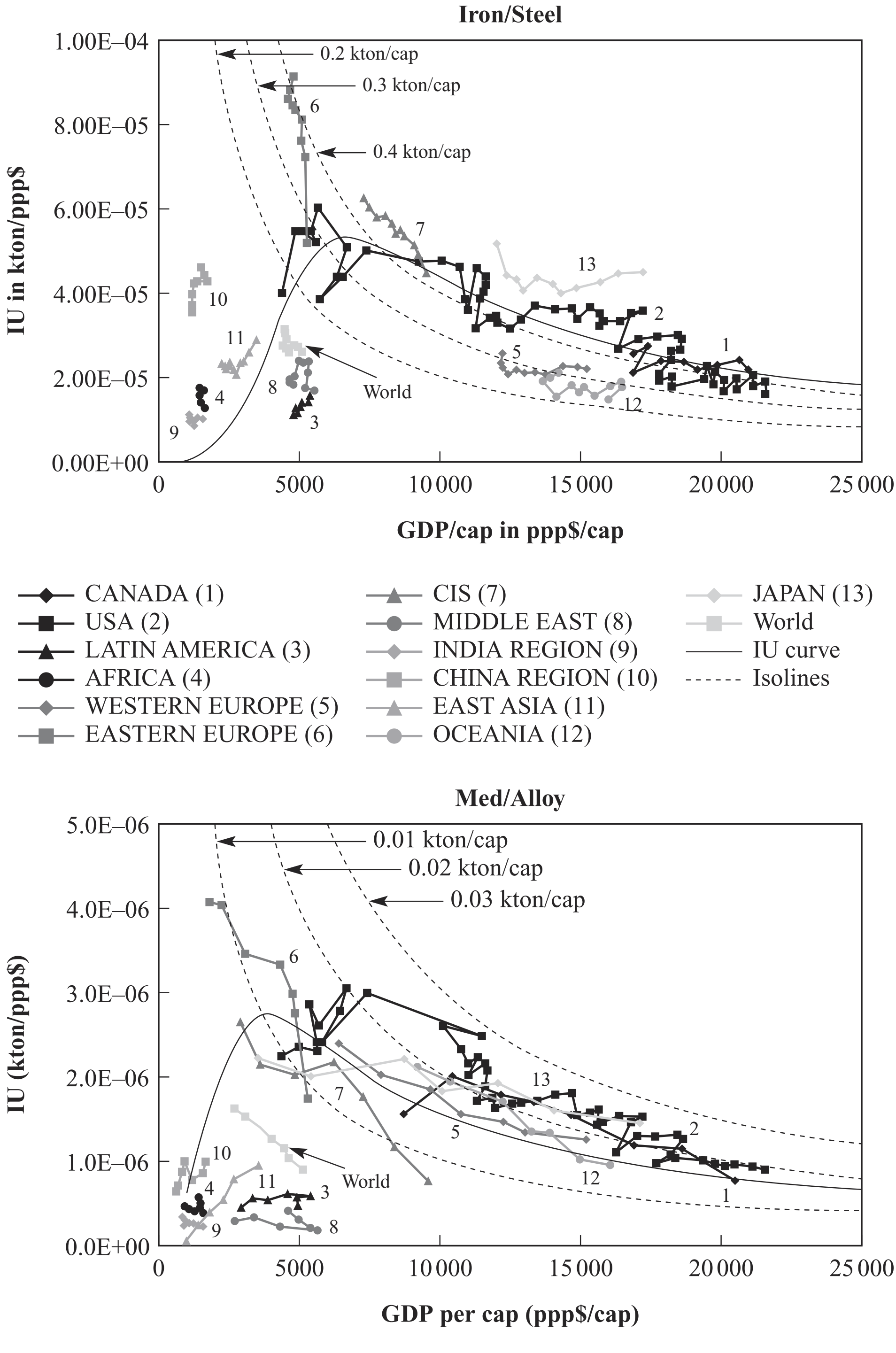


Figure 29.4 IU curve for iron/steel and MedAlloy use in 13 global regions

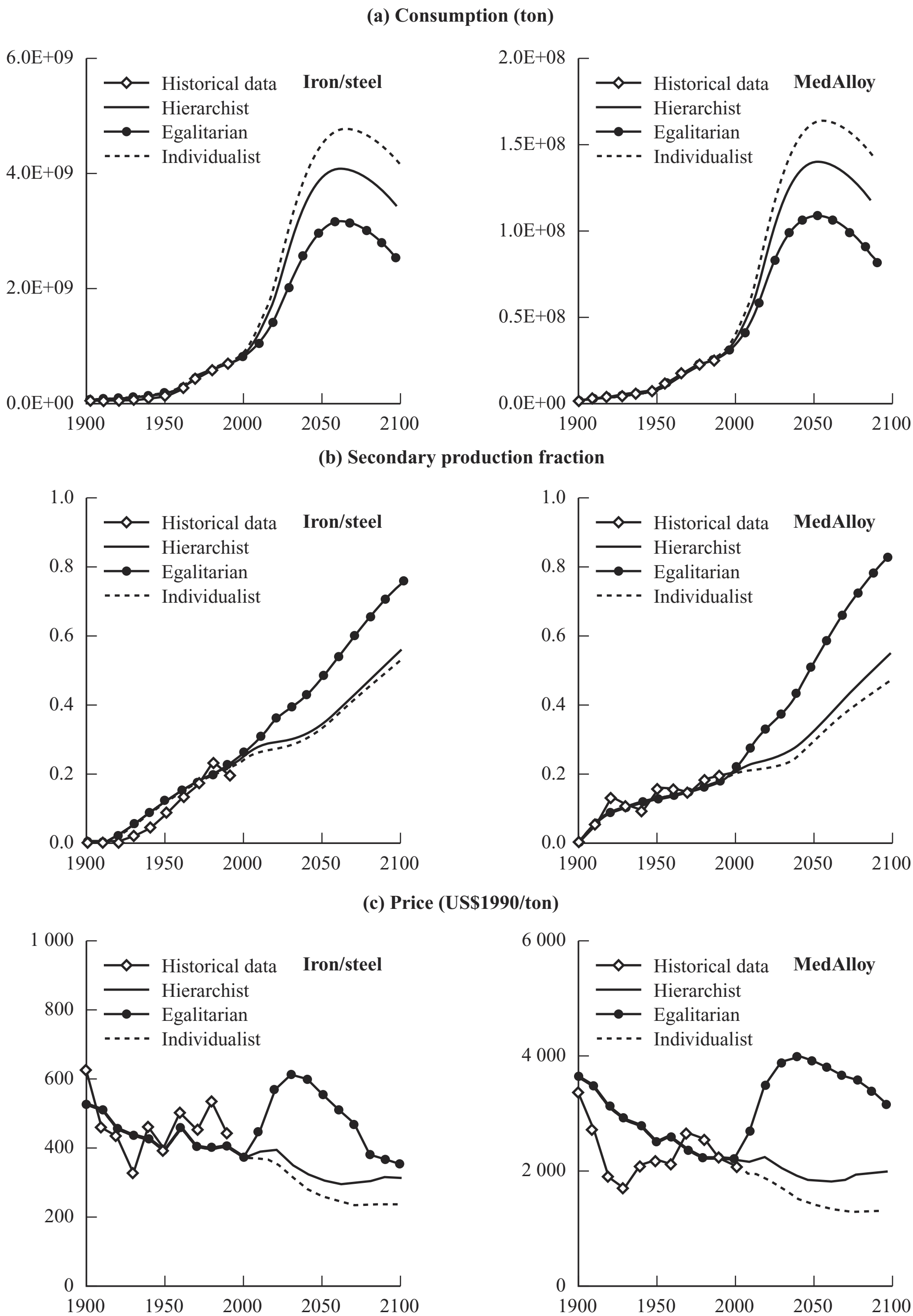


Figure 29.5 Model results, 1900–2100: (a) consumption; (b) secondary production fraction; (c) price; (d) ore grade; (e) energy consumption

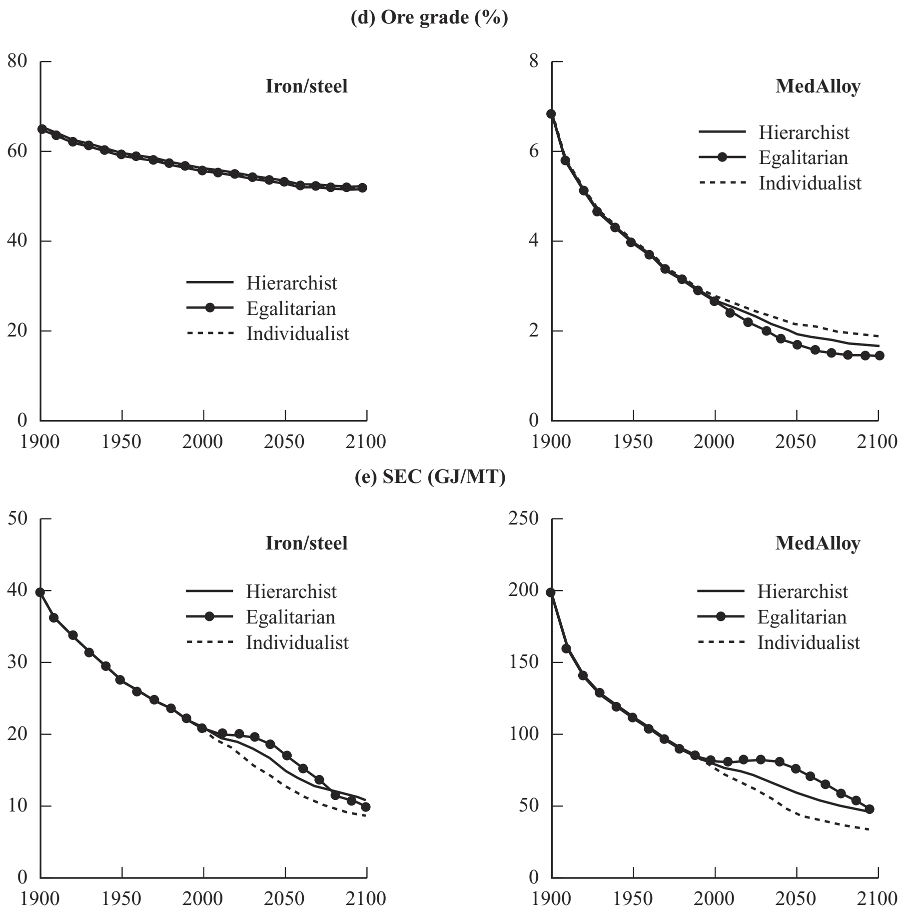


Figure 29.5 (cont.)

model has the option to scale down future metal demand from the IU curve, for example to simulate the effect of technology transfer. Finally, demand in the model is also influenced by metal prices, for which we have used a simple elasticity function. The calibration of the IU function, based on the regional use trends, produced the IU curves shown in Figure 29.4.

Metal Production

The model distinguishes between primary production and recycling (see Figure 29.1). Their market shares are calculated on the basis of relative costs: that is, if costs are equal, market shares are also equal; if relative costs rise, the market share declines (for more details, see de Vries and van den Wijngaart 1995). In contrast to the demand formulation, we have assumed that, since metal ores and commodities are traded globally, most of the production dynamics included in the model can adequately be described at the one-world level. A one-world production model also avoids the need to describe the complex dynamics of regional metal production and trade.

Primary production

Primary production in the model encompasses all processes from mining and milling to smelting and refining. With respect to its dynamics (see Figure 29.2), two main loops can be distinguished: the long-term loop describing the trade-off between depletion and learning dynamics, and the short-term demand–investment–production–price loop. Here, we focus on the first, since it is more relevant given our model objective.

Long-term loop: learning and depletion

We have defined the issue of (potential) depletion completely in terms of quality, that is, ore grade. Assuming that resources of the highest quality are exploited first, further exploitation always leads to quality decline. Such a decline can lead to increased energy requirements and production costs per unit of primary metal if not offset by further technological development. In the past, technological development has been extremely important. Considering steel, for instance, in this century we have seen the introduction of the Bessemer production process, open hearth steel production, oxygen steel production, continuous casting processing and electric arc steel production – each time reducing energy requirement and production costs. Several authors have found that technology development can be described well by a log-linear relationship between cumulative production and the efficiency of the process ('learning-by-doing'). The more frequently a process has been performed, the more knowledge has accumulated to improve its efficiency. The progress ratio of such curves (defined as one minus the factor with which production per unit of capital or energy improves on a doubling of cumulative output) generally varies between 0.65 and 0.95 (Chapman and Roberts 1983; Argote and Epple 1990; Weston and Ruth 1997). Based on historical data, a progress ratio of 0.8 is used in the metals model (Chapman and Roberts 1983).

Ore grade decline by itself has received considerable attention from geologists in terms of quantity–quality relationships. The relationship chosen for the metals model is that of Lasky, describing ore grade as a function of the cumulative tonnage of ore produced. (In fact, this empirical relationship can be looked upon as the high grade end of a log-normal distribution. Log-normal distribution of elements is often assumed to be a fundamental law of geochemistry.) Lasky-type relationships should be used with care (Brinck 1979; Singer and Mosier 1981). Deffeyes and MacGregor (1980) have derived several depletion constants for such curves that are also used in the metals model. Parameters for the capital –output ratio (COR) and energy intensity (EI) equations are based on de Vries (1989a) and Chapman and Roberts (1983).

Secondary Production

In principle, recycling limits waste flows to the environment and reduces energy requirements since the energy-intensive mining and concentration stages are avoided. The main factors influencing the recycling rates are (a) scrap availability, (b) relative processing costs of scrap and virgin metals, and (c) possibility of cost-effective scrap collection (Duchin and Lange 1994).

We distinguish between two types of recycling in the model: (i) old scrap recycling, that is, direct recycling of metal products after their use ($p=2$ in Figures 29.1 and 29.2); and (ii) dumped scrap recycling, that is, recycling of metal products after disposal ($p=3$ in Figure 29.1). Secondary production in the literature sometimes also includes 'home scrap' and 'process scrap' recycling, which refers to recycling within the metal industry. These types of recycling have not been modeled explicitly as they are considered as part of the primary production process.

Dynamics of secondary production

The main difference between the model dynamics for recycling and those for primary production is that the costs for old scrap recycling are assumed to depend on the quality of the material to be recycled. As easily recyclable fractions will normally be recycled first, recycling will become more expensive if a higher fraction of old scrap is recycled (Gordon *et al.* 1987). For both capital and energy intensity we have used the relationship of Gordon *et al.* (1987). The energy intensity of recycling of the metals included in this study is, on average, about one-fourth to one-third of primary production (Chapman and Roberts 1983; Frosch *et al.* 1997).

CALIBRATION

In order to show the relevance of the model and to find estimates for some of the model parameters, the model has been calibrated by comparing simulation results to historical data from 1900 to 1990. As previously stated, some of the historical data are uncertain. However, our objective with the model is not the exact reproduction of past trends; rather, we will concentrate on the long-term trends. This chapter will not describe the data collection and calibration efforts in detail but, instead, will be devoted to the most relevant results.

Figure 29.5(a) shows both historical and simulated consumption rates for the 1900–2100 period. The model reproduces past consumption fairly well on the basis of historical regional GDP and population data (Klein-Goldewijk and Battjes 1997) and historical consumption data from the US Bureau of Mines (USGS 1999; USBM various years), Metallgesellschaft (various years) and International Institute for Iron and Steel (IIIS 1996) and Schmitz (1979). A closer look reveals some discrepancies for both iron/steel and MedAlloy. Further improvement is only possible by also introducing region-specific IU curves (in addition to region-specific income and population trends). Figure 29.5(b) shows that simulated secondary production rates as a fraction of total production also follow the historical estimates fairly well.

In the model, the empirically derived relationships between ore grade and cumulative production result in steadily declining ore grades (see Figure 29.5(d)). We also know empirically that the ore grade for several metals has decreased considerably. At the start of the 20th century, for instance, iron ore was generally mined in the USA at grades more than 60 per cent, while nowadays grades of 20 per cent are common. In South America and Africa, in contrast, high grades are still found. For copper ore, the average ore grade in the USA was about 3 per cent around 1900 while the current grade is about 0.9 per cent (USBM 1993). In other regions grades are sometimes higher and the spread is substantial, but evidence of declining grades also exists here. It should be noted that ore grade decline is due, not only to depletion, but also to a transition to cheaper open-pit mining. Figure 29.5(d) shows that for MedAlloy the model indicates a trend from a 7 per cent global average ore grade in 1900 to a 2.8 per cent grade in 1990. For iron/steel the trend is from 64 per cent to 55 per cent. Obviously, this seriously impacts mining waste which is even aggravated by the trend to open-pit mining with its higher amounts of overburden.

Finally, the model reproduces both overall energy use of the iron and steel sector (IEA 1999a, 1999b) and energy intensity very well (Figure 29.5(e); Worell *et al.* 1997; Hendricks *et al.* 1998). Historical energy intensity for various countries between 1980 and 1991 was between 20 and 50 gigaJoules per metric ton (GJ/MT) of crude steel, while rates of efficiency improvement varied between 0.0 per cent and 1.8 per cent per year. The model shows a global energy intensity of 20–30gJ in the same period (including recycling) and an annual improvement of 0.6 per cent.

LOOKING TO THE FUTURE: A PERSPECTIVES APPROACH

The model was designed to examine long-term dynamics of metal use. Several researchers have indicated that assumptions and views on the long-term sustainability of resource use are highly dependent on people's worldview (compare Latesteijn *et al.* 1994; Tilton 1996). We have used the model to develop scenarios that reflect the main positions in the current debate. Such model-designed scenarios might contribute to the discussion as they force the debaters to provide a quantitative, internally consistent, foundation for their opinions and expectations. Developing the scenarios, we have used the cultural theory of Thompson *et al.* (1990) as heuristic for scenario development (see also Rotmans and de Vries 1997).

The two scenarios we will discuss are linked to two different schools of thought with respect to depletion of exhaustible resources: the concerned egalitarian view and the unconcerned individualistic view (Tilton 1996). The scenarios also have some assumptions in common: (a) they both assume considerable economic growth; (b) they assume global population to stabilize around 2060 and decline afterwards to 7.1 billion in 2100; and (c) they assume that current low-income regions will follow a more or less similar path in terms of IU to what high-income regions followed earlier. Because almost 80 per cent of the global population lives in low-income regions, this last assumption creates a potentially great demand for metals. Comparative results are shown in Table 29.1.

The Concerned Egalitarians

The basic assumption of the 'concerned egalitarian' view is that resources are limited, despite the fact that new discoveries and technology have increased mineral reserves in the past and are likely to do so in the future. The limits, however, are drawn by acceptable social impacts and environmental damage associated with resource production (Friends of the Earth 1998; WWF/IUCN 1999). And because population growth and spread of material-intensive lifestyles could increase demand by an order of magnitude, major policy initiatives are needed to reverse current trends. In our scenario, we have used the recently developed IPCC (Intergovernmental Panel on Climate Change) B1 scenario

Table 29.1 Model results in 2100 for three scenarios plus the egalitarian nightmare (1990 = 1.0)

	MedAlloy				Iron/steel			
	Hier.	Ega.	Ind.	Nightmare	Hier.	Ega.	Ind.	Nightmare
Absolute indicators								
demand	4.2	3.0	5.2	10.0	5.0	3.7	6.2	12.4
energy consumption	2.2	1.6	2.0	8.8	2.4	1.5	2.4	7.1
waste	4.1	1.3	5.2	17.4	3.6	1.6	4.6	10.6
ore grade	0.6	0.5	0.6	0.3	0.9	0.9	0.9	0.9
price	0.9	1.4	0.6	1.6	0.8	0.9	0.6	1.0
Relative indicators								
IU	0.20	0.19	0.20	0.18	0.24	0.23	0.23	0.23
recycling fraction	2.9	4.5	2.6	2.8	2.4	3.3	2.3	2.0
total energy efficiency	0.5	0.5	0.4	0.9	0.5	0.4	0.4	0.6
waste efficiency	1.0	0.4	1.0	1.7	0.7	0.4	0.7	0.9

Notes: Hier. = hierarchical, Ega. = egalitarian, Ind. = individualist, Nightmare = egalitarian nightmare. Waste is the sum of mining waste and depreciation flows. Prices, energy efficiency and waste efficiency are the average value for all processes (primary and secondary).

(IPCC 2000) to reflect this, representing moderate population growth and modest, but more equally distributed, economic growth. This has been combined with a relatively pessimistic view of the rate at which ore grades decline as production continues. We have also assumed that a high tax on primary production is introduced to internalize environmental costs and accelerate recycling (for example, Ayres 1997a). Finally, we have assumed that technology transfer will lower the top of the IU curve for currently developing regions. Compared to the other scenarios, metal demand in the egalitarian scenario is relatively low and a high fraction of demand is covered by secondary production (see Figure 29.5(a) and (b). In a world governed by egalitarian policies, recycling is, as expected, one of the ways in which the negative consequences in terms of energy and waste flows from low-grade ore mining can be mitigated.

The Unconcerned Individualists

The unconcerned view points out that any appropriate estimate of the ultimate stock of metals shows an enormously long lifetime. Furthermore, new technologies keep the costs of exhaustible resources falling (Simon 1980). If resource allocation is left to the marketplace, the price system will foster exploration of new resources, material substitution or even recycling (Barnett and Morse 1963). The optimism of the individualistic view is implemented in the model by low estimates for the depletion parameters and relatively rapid technological change. High economic growth is assumed, as in the IPCC A1 scenario (IPCC 2000). The saturation level of the IU curve is put at a slightly higher level to reflect material-intensive lifestyles, which results in a relatively high metal demand (Figure 29.5(a)). Despite high metal consumption, the optimistic assumptions regarding technology and depletion result in (a) a relatively low energy intensity (see Figure 29.5(e)), (b) slow ore grade decline (see Figure 29.5(d)), (3) low metal prices (see Figure 29.5(c)) and (4) little environmental waste generated per unit of primary metal produced (see Table 29.1).

General Conclusions

Both scenarios are in fact 'utopias' based on self-consistent assumptions regarding world view and human responses. Neither of them results in clearly undesirable effects in terms of production costs, energy consumption or environmental consequences. Some general conclusions can be drawn:

1. If industrializing countries follow the same pattern in terms of an increasing IU, global metal demand could increase significantly in the next century (by a factor of 2–4) and the declining trend in global IU may be temporarily reversed.
2. The abundance of iron/steel metal implies that trends in declining energy intensity and production costs simply continue. The share of steel production in total energy consumption remains at a level of between 5 and 10 per cent. In 1995, this level was 7.4 per cent (IEA 1999b).
3. For MedAlloy, historical decreases in energy and capital intensity for primary metals do not continue in the egalitarian scenario because technological innovations are offset by the relatively fast decline in ore grade.

But what happens if the world does not turn out as expected in the perspective in question (thus, is not a utopia)? Obviously, the individualist scenario is considered to be a risky adventure from an egalitarian perspective, as it assumes that depletion occurs only very slowly, technology changes fast, relatively large amounts of waste can be generated without major environmental repercussions and large differences in wealth can be sustained. The egalitarian scenario from the individualist perspective is based on a lack of courage, resulting in missed economic opportunities. Clearly, the 'government' in this scenario will lose its support if there are signs that the natural system is more robust and economic sacrifices are perceived as unnecessary.

The most obvious risk scenario is that of the 'egalitarian nightmare': the management style of the 'unconcerned individualist' scenario, plus ore grade decline and technological development expected in the egalitarian scenario. Global population, with no growth restrictions, is assumed to reach 14 billion. The 'egalitarian nightmare' results in a Table 29.1 picture: a world in which metal use generates large and increasing energy use, which in turn aggravates the impending threat of climate change; rising metal prices keep less industrialized regions in a poverty trap and enormous fluxes of mining waste put an ever-increasing stress on ecosystems, resulting in loss of biodiversity.

CONCLUSION

This chapter has described a metal model which simulates some major long-term trends in production and consumption of metals. Integrating earlier work in this area, the model's focus is on long-term dynamics and it is primarily meant as a tool for analysis, and clearly not for predictions. It aims to contribute to understanding the 'material' economy, which has been less thoroughly analyzed than energy and fuel fluxes (for example, with regard to dematerialization and resource degradation). The information can also be used for energy modeling, as metal industries consume between 5 and 10 per cent of global energy.

Our simulation experiments indicate that the model is fairly well able to reproduce the long-term trends in the 1900–1990 period. Our model-based scenarios representing the major paradigms involving unsustainable resource use allow a more quantitative interpretation of major controversies and risks involved. In all modeled scenarios, industrialization and economic growth in current low-income regions lead to a strong growth in metal demand – and even a temporary rise in global intensity of use. For abundant metals, resource quality decline is expected to be limited. For the less abundant metals, however, assumptions with regard to depletion and technology development can result in resource degradation and strong increases in energy consumption and waste production in particular, in combination with an unconcerned management style and pessimistic assumptions about the availability of high-grade ore. The differences between the scenarios are important for future policies. Obviously, not all world views and management styles can be right, but at the moment there is no evidence to rule out any of them: there is no proof that the world will soon run out of cheap metals or suffer from intolerable environmental side-effects. Nor is there any support for the complacent view that everything will sort itself out in the marketplace. Further research should be directed to resolving, as far as possible, the existing controversies.

The current model also has shortcomings. First of all, trends of IU should be analyzed in view of the underlying trends, such as substitution (for example, plastics and aluminum) and economic trends (sectoral composition of economies). This requires more integration of technical knowledge (for example, from life cycle analysis) and existing economic models. The model could also be improved by including more detail on the production side and including regional production curves.

APPENDIX: CRUCIAL MODEL EQUATIONS

This appendix displays the most crucial model equations for the minerals model discussed in Chapter 29. (Note: ppp\$ = constant purchasing power parity corrected dollars; J = Joules.)

$$D_R = IU_R \times GDP_R \times P = \left(\frac{x_1}{GDPpc_R + x_2 \times GDPpc_R^{x_3}}\right) \times F \times GDPpc_R \times POP_R \times P \tag{29A.1}$$

where D_R = metal demand per region (kg); IU_R = intensity of use (kg/ppp\$); GDP_R = real regional gross domestic product (ppp\$); x_1, x_2 and x_3 are empirically determined constants (in scenarios x_1 is varied); $GDPpc$ = real regional GDP per capita (ppp\$/capita); and POP_R = regional population (per capita). F (= 1 by default) can be used to scale down demand from historic trends in projections (see text). P describes the effect of price on demand

$$P = -elas \times [\ln(price_t) - \ln(price_{t-1})] + 1 \tag{29A.2}$$

where *elas* = price elasticity and *price* = price of metal (\$/kg) at time t.

$$Y_p = Y_{0,p} Q_p^{\;-n} \tag{29A.3}$$

Y = learning factor; Q = cumulative production (kg); n = learning constant; and p indicates the different types of production (primary/secondary).

The following three equations (29A.4 to 29A.6) are relevant for $p = 1$ only.

$$g = g_0 Q_m^{\frac{-1}{l}} \tag{29A.4}$$

where g = ore grade; Q_m = cumulative mine production (kg); and l = depletion constant.

$$EI = \epsilon_{0,SR} + \frac{\varepsilon_{0,MM}}{g} + Y \ \frac{\varepsilon_{SR} + \varepsilon_{MM}}{g} \tag{29A.5}$$

where EI = energy intensity (J/kg) and $\varepsilon_{0,SR}$ and $\varepsilon_{0,MM}$ are the energy requirements (J/kg) for smelting and refining (SR) and mining and milling (MM). A distinction is made in a minimum energy requirement (ε_0) based on thermodynamics and a part depending on technological development.

$$COR = Y\left[\gamma_1 + \gamma_2 \frac{g_0}{g} Q^{-\alpha} + \gamma_3 \left(\frac{g_0}{g}\right)^{-\beta}\right] \tag{29A.6}$$

where COR = capital–output ratio (\$/kg) and α, β, γ_1, γ_2, γ_3 are constants representing the effects of economies of scale and ore grade decline on the capital–output ratio. (Based on de Vries 1989a.)

$$costs_p = C_{en,p} + C\Omega_{,p} + C_{expl,p} = price_{en} \times EI_p + COR_p \times Ann + C_{expl,p} \tag{29A.7}$$

where $costs$ = costs (\$/kg), C_{en} = energy production costs (\$/kg), C_{cap} = capital production costs (\$/kg), C_{expl} = exploration costs (\$/kg) = 0 except for $p = 1$, $price_{en}$ = the average price of energy (\$/J) and Ann = annuity factor.

$$produc_p = D \times IMS_p \approx D \times \left(\frac{price_p^{\lambda}}{\sum_p price_p^{\lambda}}\right) \tag{29A.8}$$

where $produc_p$ = production per production category p (primary or secondary) (in kg), IMS = indicated market share, price$_p$ = price of metal produced by production category p (more or less equal to costs) (\$/kg) and λ = logit parameter). If capacity is insufficient, the real market share of category p can be lower than suggested by IMS. D = demand for metals.

Table 29A.1 Global consumption data (primary and secondary) for abundant metals and metals of medium abundance (MMT metal content)

	Fe	Al	Mn	Mg	Cr	Cu	Pb	Zn	Sn	Ni
1900	38.2	0.0	1.1	0.0	0.0	0.5	0.9	0.5	0.1	0.0
1905	53.7	0.0	1.0	0.0	0.1	0.7	1.0	0.7	0.1	0.0
1910	62.1	0.0	1.8	0.0	0.1	1.0	1.2	0.9	0.1	0.0
1915	59.5	0.1	1.4	0.0	0.1	1.2	1.2	1.1	0.1	0.0
1920	62.6	0.1	1.8	0.0	0.2	1.1	1.0	0.9	0.1	0.0
1925	75.7	0.2	2.7	0.0	0.2	1.7	1.7	1.5	0.1	0.0
1930	78.6	0.3	0.9	0.0	0.2	1.9	1.9	1.6	0.2	0.1
1935	74.0	0.3	0.8	0.0	0.3	1.7	1.6	1.6	0.2	0.1
1940	105.1	0.9	2.5	0.0	0.5	2.7	2.0	2.2	0.2	0.1
1945	85.7	0.9	1.8	0.1	0.4	2.7	1.5	1.9	0.1	0.1
1950	139.1	1.6	3.2	0.0	0.8	3.1	2.3	2.6	0.2	0.1
1955	204.2	3.2	4.8	0.1	1.2	3.7	2.9	3.3	0.2	0.2
1960	266.9	4.3	5.9	0.1	1.3	5.2	3.4	3.8	0.2	0.3
1965	374.4	7.0	6.9	0.2	2.1	6.0	4.1	4.9	0.2	0.4
1970	494.5	10.4	7.8	0.2	2.7	7.5	4.4	6.2	0.2	0.6
1975	568.1	12.2	9.5	0.3	3.4	8.2	5.0	6.7	0.2	0.6
1980	614.8	16.8	10.6	0.4	3.5	9.3	5.3	6.9	0.2	0.7
1985	616.8	19.0	11.1	0.4	3.6	9.8	5.1	7.4	0.2	0.8
1990	677.7	22.4	12.0	0.4	3.9	10.2	5.2	7.8	0.2	0.9

Source: Klein-Goldewijk and Battjes (1997), adapted and supplemented by the authors.

30. Risks of metal flows and accumulation

Jeroen B. Guinée and Ester van der Voet*

Heavy metals are key issues in environmental policy and management. Environmental problems related to heavy metals have a long history. Heavy metals, despite the fact that some metals are essential elements, have toxic properties leading to adverse effects on human and ecosystem health even in small doses. Another problem-causing property is their non-degradability: once they enter the environment they will remain there for a long time. Metals tend to accumulate in soils and sediments, with real immobilization due only to geological, and therefore extremely slow, processes. Accumulation in the food chain may lead to an increased stock in biota, thereby magnifying the human dose.

Well-known examples of metals poisoning in past centuries include the lead poisoning from water pipes in ancient Rome and the mercury poisoning of the 'mad hatters' in Europe (Markham 1994; O'Carroll *et al*. 1995). In this century we have seen, among other cases, the tragedy of mercury poisoning in the Minamata Bay in Japan through consumption of sea food, that of cadmium poisoning through consumption of polluted rice and that of arsenic in Bangladesh tube wells (Pearce 2000). Lead in paint, used extensively in many older slum buildings, has caused serious health problems in many cities, especially for children. These and similar incidents have spurred governments to implement environmental policies and industries to reduce their emissions significantly (OECD 1993c, 1994b). Comparing current emissions from industrial and other point sources to those of several decades ago, at least in the industrialized countries, there has evidently been a very major reduction (Ayres and Rod 1986; Stigliani and Anderberg 1994; VROM 1993).

Present policies regarding heavy metals include not only end-of-pipe emission reduction but also recycling and even more source-oriented measures limiting or banning certain applications altogether (for example, bans on lead in gasoline in various countries). It is generally felt that the main problems have been solved as a result of these policies and that it is now a question of tying up a few loose ends and of continuing to enforce legislation. One of these loose ends is the existence of polluted sites, a relic of the past, described as 'chemical time bombs' by Stigliani and Salomons (1993). Such sites may become unsuitable for agriculture or housing construction. If they remain unattended, metals may become available and leach to the groundwater through increasing soil acidity. Other loose ends refer to applications considered risky but not (yet) regulated, such as some metal-based pesticides and paints. As shown in Table 30.1, however, it also appears that global primary production rates of most metals remain rather constant or slightly decrease (cadmium and lead) or even increase (copper and zinc).

These (constant or increasing) global production rates have to result in either (constant or increasing) emissions to water, air and soil or accumulation in the economy, that is in

* This chapter is largely based on Guinée *et al*. (1999) and van der Voet *et al*. (2000a).

Table 30.1 Global production rates of some metals for the period 1980–92 (kMT/yr)[1]

	Average 1985–9[2]	1995[3]	1996[4]	1997[4]	1998[5]	1999[5]
Cadmium[6]	20	19	19	19	20	20
Copper	8342	9800	11000	11300	12200	12600
Lead	3401	2800	2920	2900	3100	3040
Zinc	7001	7070	7440	7800	7550	7640

Notes:
[1] Data cover intentional metal ore production unless noted otherwise.
[2] USBM 1992 (Minerals in the World Economy).
[3] US Geological Survey, *Mineral Commodity Summaries*, January 1996 (*http://minerals.usgs.gov/minerals/pubs/mcs/*).
[4] US Geological Survey, *Mineral Commodity Summaries*, January 1998 (*http://minerals.usgs.gov/minerals/pubs/mcs/*).
[5] US Geological Survey, *Mineral Commodity Summaries*, February 2000 (*http://minerals.usgs.gov/minerals/pubs/mcs/*).
[6] Cadmium extracted from zinc ore.

Source: van der Voet *et al*. (2000b).

capital goods, intermediate products, consumer goods and wastes. If metals are accumulating in the economy, emissions may be only temporarily decreasing and may well increase in the future, for example through corrosion, inadequately controlled incineration and landfill of municipal and industrial solid waste (cf. Ayres and Rod 1986; Stigliani and Anderberg 1994; Bergbäck and Lohm 1997). Individual countries may also export these metals to foreign countries and thus shift the problem abroad.

The emission reductions mentioned above may be a temporary trend rather than a sustainable solution and the feeling that the main metal problems are solved may be premature. It is thus useful to analyze heavy metal flows and accumulations in the economy and environment in an integrated way. For this, substance flow analysis or material flow analysis are available as methods. In the next section, a short reference to these methods and related models will be made and some useful indicators will be discussed. Then results of a Dutch project including three case studies applying integral analyses methods are discussed. In the final section, conclusions on risk management of heavy metals are drawn.

LINKING FLOWS AND ACCUMULATIONS: METHODS, MODELS AND INDICATORS

Frequently applied methods for integral analysis of flows and accumulations of substances through economy and environment are substance flow analysis (SFA) and material flow analysis (MFA). In this chapter the focus is on SFA. For a description of MFA, see Chapter 8.

SFA offers a consistent and integral description of material or substance stocks within and flows within and between the economy and the environment in a certain time period, for a certain region. It is based on physical input–output analysis in which the materials balance principle holds for each economic or environmental sector. SFA can be implemented through modeling. Various models of economic flows and stocks have been developed, such as FLUX (Boelens and Olsthoorn 1998; Olsthoorn and Boelens 1998), STOCKHOME (Bergbäck *et al.* 1998) and a dynamic copper model (Zeltner *et al.* 1999). Other models focus on modeling environmental flows and stocks such as DYNABOX (Heijungs 2000) and Dynamic (Soil Composition) Balance (D(SC)B) (see Chapter 33).

Bookkeeping, steady state and dynamics are important concepts in modeling (see Chapter 9). In the case study presented below, the steady-state concept is of particular importance. As explained in Chapter 9, the outcome of this type of modeling provides no prediction but an assessment of the long-term sustainability of a certain substance regime compared to the present situation The steady-state situation of, for example, the 1990 flows and stocks of a substance indicates the eventual magnitude of these flows required to maintain the 1990 regime indefinitely.

Material and substance flow analysis (MFA/SFA) studies are designed to support environmental decision making. Although many SFA/MFA case studies have been carried out successfully, the issue of connecting such research with policy has arisen frequently (Brunner and Lahner 1998). Researchers may feel the results of SFA/MFA studies are giving clear messages, but for policy makers the implications are often not quite so self-evident. Several authors have attempted to define indicators in order to bridge this gap (van der Voet 1996; van der Voet *et al.* 1999; see also Ayres 1996; Azar *et al.* 1996; Wernick and Ausubel 1995). Possible indicators include the following:

- indicators for the fate of mined heavy metals (total emissions; total landfill; accumulation in the economy; pollution export);
- indicators for the fate of mined heavy metals (total emissions; total landfill; accumulation in the economy; pollution export);
- indicators for evaluation of present management in terms of sustainability (environmental risk ratio (PEC/PNEC) and transition period; human risk ratio (PDI/TDI) and transition period; environmental accumulation; depletion rate);
- indicators for design of a sustainable management regime (technical efficiency; recycling rate; use level; economic dissipation; disturbance rate).

For detailed descriptions, see van der Voet *et al.* (1999).

CASE STUDIES

In this section, we summarize case studies using three of the indicators listed above: the total emissions and two-risk ratio (and transition period) indicators.

1. *Total emissions* This indicator gives the aggregate emissions (mass/yr) from the economy to the different environmental media (air, water, agricultural and non-agricultural soil). It indicates environmental pressure and it is an early warning for the steady-state risk ratios. It may be compared with emission targets for each medium.
2. *Risk ratio and transition period* The risk ratio (dimensionless) is calculated for human toxicity and aquatic and terrestrial ecotoxicity, being the daily intake or the environmental concentration in a medium divided by the acceptable/tolerable daily intake or concentration standard, respectively, for that medium. It indicates the

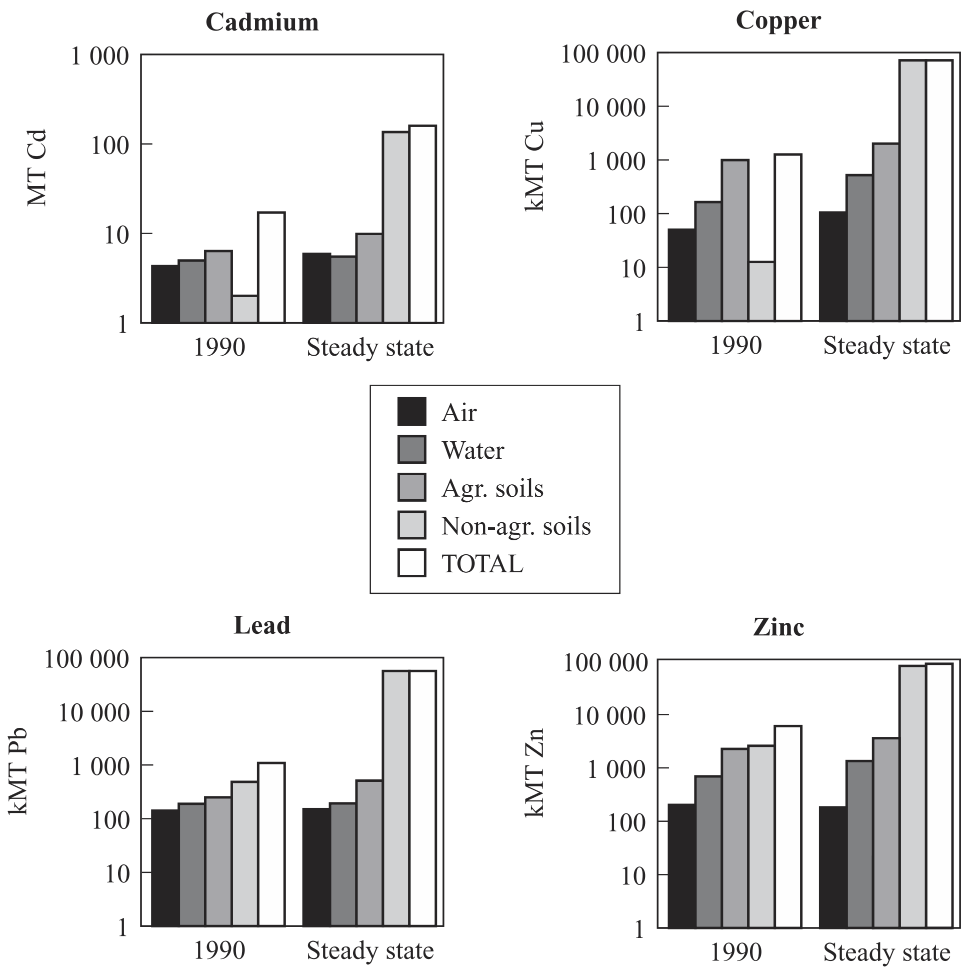
potential risk (values > 1) emissions pose to human and ecosystem health. This indicator can be calculated for a base year (for example, 1990) by using empirically measured concentrations, for the steady-state situation by using a steady-state SFA/ environmental fate model or for a year in the future by using a dynamic model: for example, DYNABOX (Heijungs 2000), USES-LCA (Huijbregts 2000) or the Dynamic Soil Composition Balance model (see Chapter 33).
3. *Transition period* The time it takes for the risk ratio to equal 1.

The contrast between decreasing emissions of the metals cadmium, copper, lead and zinc and continuously increasing input into the economy has been analyzed in three case studies for the Netherlands. The main research questions concerned the fate of the mined metals, the link to environmental risks and ways to render the metals management regime more sustainable. Flows of metals through, and their accumulation within, the economy and the environment were quantified for 1990 in a hypothetical steady-state situation for the Netherlands as a whole, for the Dutch housing sector and for the Dutch agricultural sector. To this end, the SFA method has been applied for copper, zinc, lead and cadmium with the help of several models. See Boelens and Olsthoorn (1998), Olsthoorn and Boelens (1998), Heijungs (2000) and Chapter 33 for detailed descriptions.

The results for the total emission indicators in Figure 30.1 show, for almost all media and all metals, a strong increase of emissions in the steady-state situation compared to the 1990 situation. The increase of air emissions in the steady-state situation compared to the 1990 situation is generally moderate. The increase for cadmium is apparently due to the incineration of spent NiCad batteries. The increase for copper is due to frictional wear from overhead railway wires. Air emissions for zinc for the steady-state decrease compared to 1990, since the amount of zinc used for galvanized iron is decreasing. Besides emissions, transboundary pollution via air from foreign countries is an important source for the total input to air for all four metals; however, this source is not included in the emissions indicator.

For all four metals, the increase of water emissions in the steady-state situation compared to the 1990 situation is due mainly to the corrosion of metals in building materials (for example, zinc gutters, galvanized steel, tapwater heating equipment and bulk materials such as concrete). However, with respect to the total input to water, it is not emissions within the Netherlands but inflow of metals from outside the Netherlands that constitutes the dominant source for all four metals (in some instances over 70 per cent).

The increase of steady-state emissions to agricultural soils compared to 1990 emissions is significant for all metals and is due to increasing flows of organic manure and of source-separated vegetable, fruit and garden waste (the latter being less relevant for lead). The main source for cadmium is the continuous (and constant) inflow of phosphate fertilizer. Main source for copper and zinc is the increasing content of copper and zinc in organic manure. The ultimate source behind these increasing contents of copper and zinc in organic manure is animal fodder. It appears that in the steady-state situation the agricultural soil emissions of copper and zinc are due overwhelmingly (about 80–90 per cent) to the addition of copper and zinc, respectively, to fodder. This is an example of what might be called 'closed loop accumulation' (CLA): copper and zinc are added to fodder, which is imported from abroad and fed to Dutch cattle. The manure produced by the cattle, including its copper and zinc content, is spread on agricultural land as an organic



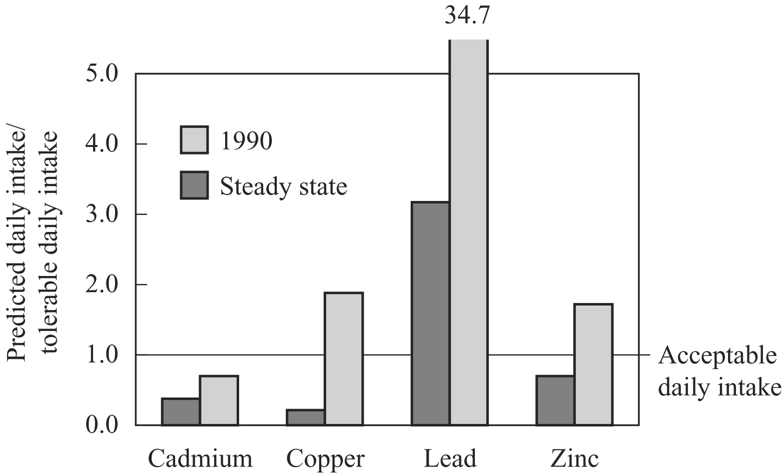
Source: van der Voet *et al*. (2000b).

Figure 30.1 Emissions of heavy metals in the Netherlands, 1990, and steady state

fertilizer. Soil concentrations of copper and zinc consequently rise and, with them, the copper and zinc concentrations in maize, pit grass, fresh grass and hay. The livestock are additionally fed with maize, pit grass, fresh grass and hay, and the metals are thus returned to the economy. The eventual steady-state soil concentration due to this cycling of copper and zinc leads to several risk ratios above 1.

The increase of steady-state emissions compared to 1990 emissions is most noticeable in the case of non-agricultural soil. For all metals, the increase of emissions at steady state is dominated by emissions from landfill sites. In a steady state the outflow equals the inflow. Since it is assumed that emissions to non-agricultural soil are the only outflow from a landfill, the emission to non-agricultural soil at steady state will equal the inflow at steady state. In the end any leakage to the environment from a waste storage site may lead to a non-sustainable situation, but the time lag may be very long (up to thousands of years). For the risk ratios, however, emissions to non-agricultural soil make a contribution of only about 10 per cent and are thus not a major source, even in the steady-state situation.

Figure 30.2 shows the risk ratios for human toxicity and aquatic and terrestrial ecotoxicity. Acceptable daily intake (ADI) values defined by the WHO and tolerable daily intake (TDI) values similarly defined by Vermeire *et al.* (1991) and Cleven *et al.* (1992) have been applied in calculating the risk ratio for human toxicity. For human toxicity the risk ratio of lead is already above 1 for 1990 and the risk ratio for the steady-state situation is above 1 for lead, zinc and copper, in decreasing order of magnitude.

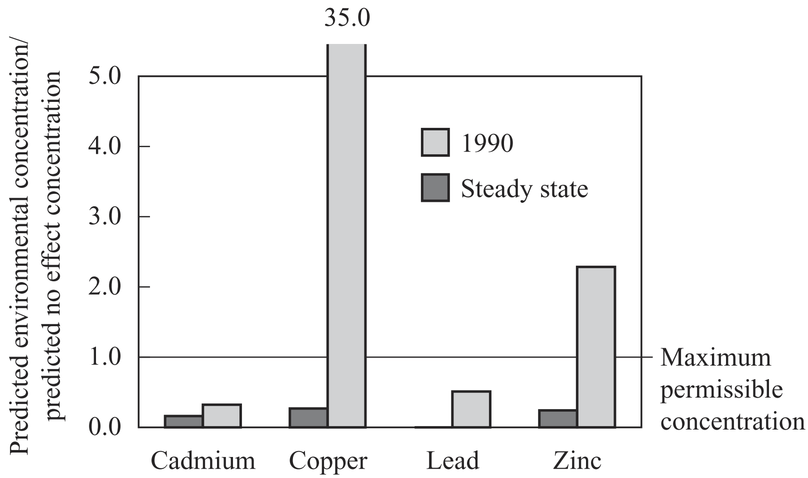


Source: van der Voet *et al.* (2000b).

Figure 30.2 Human toxicity risk ratios for cadmium, copper, lead and zinc in the Netherlands, 1990, and steady state

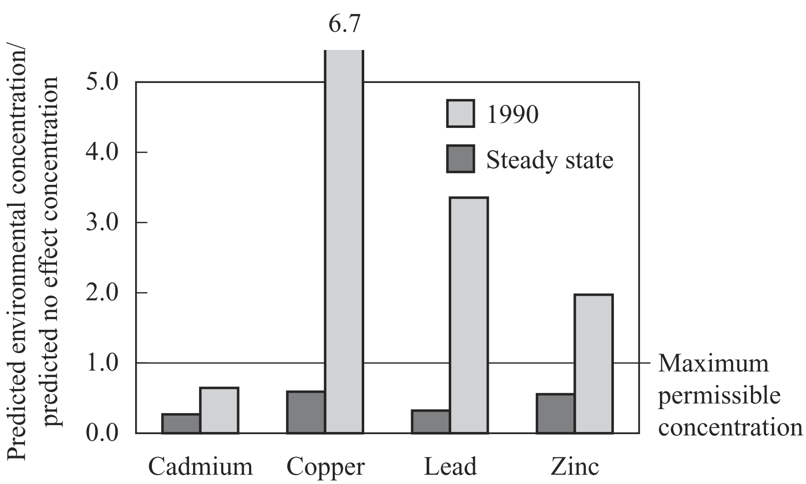
Figures 30.3 and 30.4 show the risk ratios for aquatic and terrestrial ecotoxicity. The Dutch maximum permissible concentration (MPC) standard has been applied in calculating the risk ratios for aquatic and terrestrial ecotoxicity. The MPC is defined as the sum of the maximum permissible addition (MPA) and the existing background concentration in the Netherlands, with the MPA defined as the amount of a metal originating from anthropogenic sources that is allowed on top of the natural background concentration. The MPC is an ecotoxicological value (Crommentuijn *et al.* 1997). For aquatic and terrestrial ecotoxicity it appears that copper gives the highest risk ratios, then lead and zinc, and then cadmium. These results mean that the current metabolism of these metals is generally not sustainable.

The transition periods for the various metals are shown in Table 30.2. In calculating the transition periods, current background levels in the various environmental media have been taken into due account. The transition periods vary from 0 years for cadmium in water to reach the reference and the limit value, to 1000 years for lead in water to reach



Source: van der Voet *et al.* (2000b).

Figure 30.3 Aquatic ecotoxicity risk ratios for cadmium, copper, lead and zinc in the Netherlands, 1990, and steady state



Source: van der Voet *et al.* (2000b).

Figure 30.4 Terrestrial ecotoxicity risk ratios for cadmium, copper, lead and zinc in the Netherlands, 1990, and steady state

Table 30.2 Transition period for risk ratios for cadmium, copper, lead and zinc in the Netherlands (years)

	Cadmium	Copper	Lead	Zinc
MPC terrestrial	infinite	30	550	120
ADI	infinite	460	0	130
MPC aquatic	infinite	3	1 000	16

Source: van der Voet *et al.* (2000b).

the limit value. The results for soil have been compared with the results of the more sophisticated D(SC)B model for cadmium and appeared to be fairly similar (see Chapter 33). The results for human toxicity and aquatic ecotoxicity could not be compared with any other model.

The results of these case studies on cadmium, copper, lead and zinc for the Netherlands are supported by results from similar case studies for other regions. Steady-state calculations for lead in Denmark found similar results for the emissions indicators (Hansen and Lassen 1997). (Hansen and Lassen did not calculate risk ratios and transition periods, as far as we know.) In a case study of lead and zinc in the Vienna area, measured concentrations in air, water and groundwater appeared to be mostly within established environmental standards, but metal stocks in economic goods and capital were quite large (Obernosterer *et al.* 1998). Bergbäck *et al.* (1998) in their study for the Stockholm area showed that metals continuously increase to accumulate in economic goods and capital. In the Dutch study these (increasing) stocks and relatively low environmental concentrations were the starting point for integral economy–environment modeling in a steady-state perspective. The massive economic stocks then appeared likely to cause significant future steady-state concentrations and associated high-risk ratios.

RISK MANAGEMENT FOR METALS

While emissions of metals are decreasing, their input into the economy is still increasing. What happens to this input, now and in the long run? These questions can be addressed through integrated modeling of metal flows within both the economy and the environment. Several case studies applying such integrated modeling showed that the general feeling that the metals problem is solved is premature.

Three main approaches for enhanced metals management now seem feasible: (a) the input into the economy can be lowered; (b) the output can be delayed; and (c) the output can be controlled or sequestered. The first main approach, lowering of the input, can be achieved by replacement of metals in functional applications (for example, PVC for zinc gutters), by recycling or increasing the lifespan of metals with an elastic supply (for example, Cu, Pb or Zn) and by reducing inflows as contaminants, for example in phosphate fertilizer or fossil fuels. The second main approach, delaying the output, can be achieved by keeping non-functional metals within the economy (for example, fly ash in concrete or roads). This option offers time for further development of the third main approach: control of the output, which can be achieved by physicochemical immobilization of the waste flow (for example, vitrification), by waste disposal outside the biosphere, and by bypassing the sensitive environmental routes (for example, reburial in old mines). A number of scenario calculations have been performed in order to assess the effect of several possible policy measures for zinc and copper (van Oers *et al*. 2000). Although the dynamic model used for these calculations is too unreliable for 'realistic' results, the results of the scenario calculations for zinc and copper indicate that very stringent measures are needed (for example, significant change of agricultural practice, replacement of building materials, adoption of not yet existent waste management techniques) for a sustainable metals management (van der Voet and van Oers 2000).

Although the models used include the full spectrum of flows and accumulations in the economy and environment, the results presented above for the Netherlands are merely indicative. Besides general uncertainties attached to economy environment modeling, a basic limitation is that resource availability has not been taken into account in the steady-state model used; this is in fact assumed to be infinite. So the high-risk ratios will probably not be reached because of enforced declines in resource extraction. However, this is by no means certain, given continually rising estimates of resource availability. Consequently, the results at least imply a warning signal as to the sustainability of current metal metabolism.

31. Material constraints on technology evolution: the case of scarce metals and emerging energy technologies

Björn A. Andersson and Ingrid Råde*

The evolution of the biosphere has been constrained by the relative availability of the chemical elements, and all living matter mainly consists of the abundant elements, carbon, hydrogen and oxygen. The evolution of the industrial system, too, is constrained by the availability of elements as well as the sustainable use of them (Holmberg and Karlsson 1992; Karlsson 1996; Holmberg *et al.* 1996; Azar *et al.* 1996).

A key component of every industrial ecosystem is the energy system. The growth of the world population to nine or ten billion towards the second part of the 21st century and continued economic growth will immensely increase the demand for energy services. At the same time, the carbon dioxide emissions inherently linked to current fossil fuel energy technologies need to be reduced substantially over the century. Hence, there is a need for development and large-scale growth of a range of new technologies in the energy sector.

Some emerging technologies that are promising in the short term may, however, be constrained in the long term by their requirement for scarce metals. Constraints may materialize in the form of resource scarcity or detrimental environmental effects. In this chapter we discuss such long-term material constraints and the role they may play in forming or disrupting sustainable technology trajectories. We outline some implications of the assessment for policy and strategy as well as for the scope of industrial ecology. Throughout the chapter we give examples from three promising technology domains: batteries and fuel cells for electric-drive vehicles (EVs) and thin-film solar photovoltaics (PV).

THE SCALE OF MATERIAL CONSTRAINTS

PV and EVs currently supply minor niche markets for electricity and transport but their roles in the energy system may change drastically later in this century. The current stock of installed PV capacity in 1999 was less than 1GWp (billion watts of peak power) which probably produced less than 1TWh/yr (terawatt hours). This can be compared to scenarios developed by IIASA and WEC (2000), where the energy supplied by direct solar technologies varies between 5000 and 22000TWh/yr in 2050 (that is, between 2 per cent and 7 per cent of total supply). By 2100, the PV output could be between 23000 and

* This chapter is based on the case studies in Andersson *et al.* (1998); Andersson (2000); Andersson and Jacobsson (2000); Andersson and Råde (2001); Råde and Andersson (2001a, 2001b). For more elaborate general discussions see Andersson (2001) and Råde (2001).

127 000TWh/yr (that is, between 4 and 25 per cent of total supply). There are other (non-PV) solar technologies, but the numbers imply a potential growth of PV capacity by three to five orders of magnitude. Assuming relatively high capacity utilization, the creation of a 25 000TWh/yr PV system, over the course of a century, would require an annual addition of 160GWp or 1000 times the manufacturing volume in 1998.

The stock of EVs in the world in the late 1990s was between 10 and 20 thousand units, while the number of motor cars was 500–600 million. Given a reasonably high economic growth, and no major trend break in personal mobility, we may approach a situation at the end of the century where the average car ownership in the world is about the current average in Western Europe (Azar *et al.* 2000; Schafer and Victor 2000). With a population of nine to ten billion people, this implies a total stock of four to five billion cars, which are by many envisioned to be EVs powered by batteries and fuel cells.

The question arises: will some PV, battery or fuel cell technologies which are promising in the short run be constrained by metal scarcity somewhere on the path from small-scale to large-scale implementation? To get some idea of levels, we use medium optimistic assumptions for metal requirement per utility service unit together with metal reserves to calculate the material-constrained stock (S_{MC}) for two solar cells, three batteries and one fuel cell (Table 31.1). The PV designs are constrained to produce 150 and 300TWh/yr, the batteries to power 10 to 300 million EVs and the fuel cell to power 1.4 billion EVs. The material-constrained stocks are dramatically larger than current solar electricity production and EV fleets, but significantly smaller than the envisioned demand. For an assessment comprising four types of thin film PV designs, see Andersson (2000, 2001) and Andersson and Jacobsson (2000), for nine types of batteries see Andersson and Råde (2001), and for two types of fuel cells see Råde and Andersson (2001a).

The material-constrained stocks in Table 31.1 are only an indication of scale and a point of departure. Apparently, the requirement of scarce metals is an issue for all these technologies but the many factors determining metal requirement and availability introduce a wide range of uncertainty. The actual potential may be larger as well as smaller.

CRITICAL ASPECTS OF METAL REQUIREMENT

Several factors govern the metal requirement per utility service unit of a technology: requirement of capacity per utility service unit, net metal intensity per capacity unit and efficiency of the material recovery system.

Capacity per Utility Service Unit: System Efficiency, Location and Design

The electricity produced per watt of peak power (Wp) of installed PV capacity depends on the technical efficiency of system components such as current converters as well as on system location: for example, whether the modules are installed in Algeria or northern Germany (the annual insolation varies by a factor of two). It also depends on whether their tilt angle is optimized for electricity production or for architectural expression. Mainly depending on location, the PV capacity required to generate 1kWh/yr would typically vary between 0.5 and 1Wp.

Similarly, technical efficiency parameters and design choices – such as aerodynamics,

Table 31.1 Indication of the material-constrained stock for selected technologies

Broad technology Technology design Metal	Metal requirement per utility service unit[1] (F)	Reserves 1999[2] (R)	Material-constrained stock (S_{MC} = R/F)	Metal's share of target cost[3] (%)
PV	(g/MWh/yr)	(Gg)	(TWh/yr)	–
CIGS				
Selenium	40	70	2000	0.02
Gallium	5	110	20000	0.08
Indium	30	10	300	0.3
CdTe				
Cadmium	130	600	5000	0.009
Tellurium	130	20	150	0.3
Batteries	(kg/vehicle)		(million EVs)	
Li-metal				
Lithium	10	3400	300	6
Vanadium	30	10000	300	11
NiCad				
Nickel	60	46000	800	13
Cadmium	60	600	10	4
Lead acid				
Lead	300	64000	200	10
Fuel cell				
PEMFC				
Platinum	0.025	35	1300	32

Notes:
[1] Medium optimistic assumptions: cumulative losses in material systems 10%; PV: balance of system efficiency 80%, insolation 1800 kWh/m²/yr, layer thickness 1 μm (CIGS), 2μm (CdTe), module efficiency 14% (CIGS), 12% (CdTe), Ga:In molar ratio 1:3, see also Andersson (2000). batteries: mid-values of range given by Råde and Andersson (2001b); fuel cell: 50 kWp fuel cell and 0.45g Pt/kW.
[2] *Source*: USGS (2000) and 45% Pt in platinum group metal reserves.
[3] PV cost target: 1USD/Wp module and area-related balance of system costs (Zweibel, 1999); EV battery cost target (USABCs commercialization target): 150USD/kWh (Davis, 1999); fuel cell: cost target 20 USD/kW.

vehicle size, driving range and acceleration – govern the required energy storage and peak power of an EV. The battery energy capacity required for a pure battery EV would typically fall in the range 10–40kWh, depending on vehicle technology and required driving range. The battery capacity required for a hybrid vehicle such as a Toyota Prius would be a factor of 10 lower, while the power capacity of the fuel cells would fall in the range 20–85 kWp.

Net Metal Intensity

By 'net metal intensity' we denote the mass of metal contained in the product per capacity unit. For PV, it depends on the module efficiency and the thickness of the semiconductor layer, for batteries, on the specific capacity of the metal (mAh/g) and the battery voltage, and for fuel cells on the power intensity (W/cm²) and the metal loading (mg/cm²).

Through technological development we may affect these parameters and decrease the net metal intensity (that is, dematerialize the technology; see Chapters 16 and 18).

However, there are theoretical as well as practical limits. The efficiency of thin film PV modules currently produced is within a factor of 2–4 of theoretical limits. The thickness of some thin films could in theory be reduced by a factor of 5–10 (Zweibel 1997; Andersson 2000). The net metal intensity of state-of-the-art batteries is within a factor of 2–6 of theoretical limits (Råde and Andersson 2001b). The net platinum intensity of polymer electrolyte membrane fuel cells (PEMFCs) has, over the past two decades, already decreased by two orders of magnitude. However, the remaining dematerialization potential is probably more limited (Råde and Andersson 2000a). As compared to, for example, information technologies (Allenby 1999a), the limits of dematerialization for these energy technologies are not far away.

In some cases it is possible to decrease the intensity of an element by substitution. For example, gallium may substitute for indium in a copper–indium–gallium–selenium (CIGS) PV cell. Other transition metals could possibly substitute for ruthenium in dye-sensitized PV cells and the compositions of metal hydrides in nickel–metal–hydride batteries can be altered. Of course, at some degree of substitution it is not the same technology anymore.

There are also trade-offs between various degrees of freedom, for example between film thickness and efficiency of PV cells and between cell area and the efficiency of fuel cells. Altogether, in practice, net intensities of most PV, battery and fuel cell metals could probably be decreased in the range 1.5 to 4 times from current state of the art.

Efficiency of the Material Recovery System

Losses in manufacturing and recycling add to the net metal intensity. The key parameters are losses in manufacturing and reprocessing, the utilization of manufacturing residues, collection of used products and the product lifetime. For example, suppose a logistic penetration rate of a technology extended over a century, with a product lifetime of five years (realistic for some EV batteries). In this case, a 10 per cent total loss rate in one manufacturing and recycling loop would just about double the cumulative metal requirement (Andersson and Råde 2001). However, for lead acid batteries, manufacturing and reprocessing systems exist that can keep lead losses to as little as 0.1 per cent (Karlsson 1999). Supposing 99.9 per cent of old batteries are collected, the cumulative losses would not add more than about 1 per cent to net requirements.

For the diffusion of PV technologies, with target lifetimes as long as 30 years, the utilization of materials in manufacturing, including recovery of manufacturing residues, will for a long time be of greater importance than recovery from spent modules. Processes to recover metals exist or are under development for most batteries, fuel cells and thin film PV designs. However, some of these processes are not compatible with 'true' (or 'closed loop') recycling. As an example, nickel used in nickel–cadmium (NiCad) batteries is currently not recycled to yield nickel of a grade suitable for battery manufacturers but is downgraded to ferronickel for stainless steel production (Acurex 1995). From the perspective of metal availability for a *specific* technology, such downgrading is equivalent to no recycling. Over a century, this could multiply the cumulative metal requirement by a factor of 10 (based on the assumptions given above).

In practice, there are trade-offs between net metal requirement and material system efficiency. For example, in the case of batteries, a reduction of metal intensity may reduce the lifetime, and a more complex material composition in order to improve performance may decrease recyclability.

AVAILABILITY OF SCARCE METALS

The reserves used in Table 31.1 are defined as the demonstrated resources that are economically recoverable at today's prices and extraction costs. They are by no means a definitive measure of available primary resources. Also there is another stock of recoverable metals that has been extracted in the past and still remains in the technosphere (secondary resources). Moreover, the diffusion of a technology may be constrained, not only by the available stock of resources, but also by the rate at which these are recovered and by competition for metals from other end uses. All of these factors must be considered.

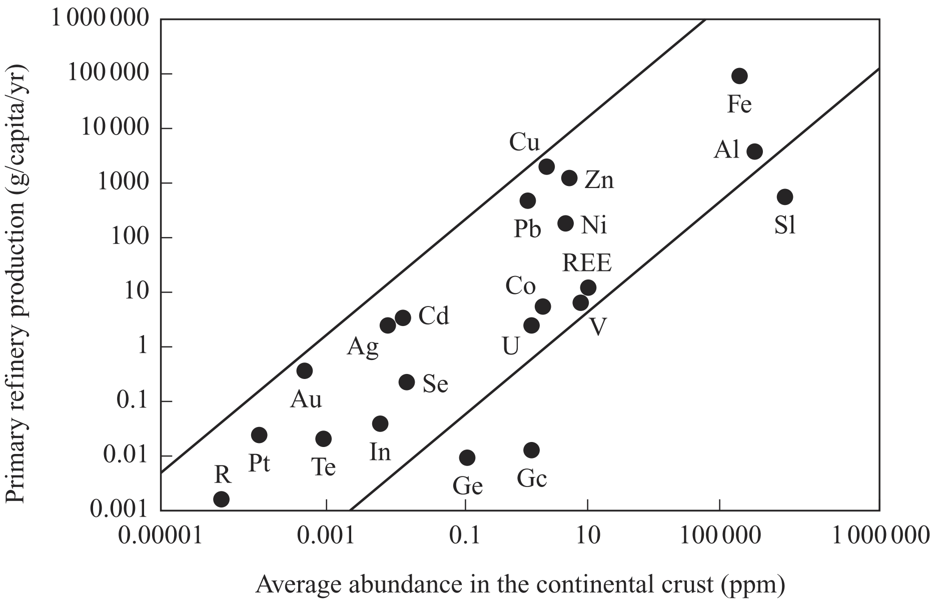
Primary Resources and Recovery Rates

There has been considerable controversy over assessments of future supplies of primary resources; see, for example, Tilton (1977, 1996). Even if there is no consensus on how to construct reliable economic indices of scarcity (Cleveland and Stern 1999), no economic indices, such as real prices or recovery costs, have clearly indicated an increasing scarcity of metals (Barnett and Morse 1963; Krautkraemer 1998). Over a large part of the 20th century, technological progress, new discoveries and substitution have compensated for the depletion of high grade ores, and kept prices from rising.

However, this by no means implies that metals are not scarce. Of prime concern for technology evolution is that the abundance of metals varies by many orders of magnitude (Figure 31.1). Iron, for example, is half a billion times more abundant in the Earth's crust than ruthenium. This difference in abundance is reflected in the industrial metabolism. The global refinery production of iron in 1999 was 90kg per capita or 50 million times that of ruthenium, which had a refinery production of less than 2mg per capita. Clearly, scarcity militates against building bridges of ruthenium.

The availability of the metals considered for batteries, fuel cells and thin film PV may be constrained in different ways. Different factors govern the availability of metals that are mined primarily as main products or high-value co-products versus those mined as low-value by-products. We may also distinguish between metals that have low and high extraction-to-reserve ratios (Table 31.2). Some main product metals such as lead, copper, zinc and nickel are mined in large quantities and have reserve-to-extraction ratios well below 100 years. These appear to be constrained by the available stock of economic resources rather than by the extraction rate.

Historically, new discoveries of metal deposits have kept adding to known resources. However, as pointed out by Skinner (1976, 1979, 1987) there is a distinct difference between abundant metals such as aluminum and iron and rare metals such as copper, zinc, nickel and lead. The former have an average crustal abundance high enough to form separate minerals in common rocks. They are mined from deposits where they have been enriched (by geological processes) by factors of two to 10.



Source: Wedepohl (1995); USGS (2000).

Figure 31.1 Metal abundance in the Earth's crust and in society

Table 31.2 Current and historical extraction compared to the reserves

	Reserves-to-extraction ratio[1]	Cumulative extraction-to-reserves ratio[1,2]
	(yr)	(–)
Lead	21	3.8
Cadmium	30	1.4
Nickel	40	0.8
Indium	42	0.4
Selenium	56	1.0
Tellurium	180	0.7
Platinum	210	0.1
Lithium	230	0.09
Vanadium	250	0.12
Rare earth elements	1400	0.014
Gallium	1500	0.008

Notes:
[1] 'Extraction' denotes refinery production for selenium, gallium, indium, cadmium and tellurium and mine production of others (see USGS 2000).
[2] See Andersson (2000), Andersson and Råde (2001), Råde and Andersson (2001a) and Sternbeck (1998).

By contrast, the rare metals have much lower crustal abundance and consequently require far greater geological enrichment to form minable deposits of separate minerals such as metal sulfides. In mined ores the rare metals are enriched from a hundred to some thousand times above their average concentrations in common rock. There is only a tiny fraction of the metal in the Earth's crust that is contained in such rich deposits (Skinner 1976). By far the major fraction of the total amount in the Earth's crust is contained in solid solutions in the silicate matrix of common rock, from which metal extraction would require immense energy inputs. Skinner (1987) has estimated the maximum land-based copper resources in mineral deposits to be in the range 1–10Pg. In fact, the known conventional resources of copper (the reserve base) plus the cumulative mine production already exceed 1Pg. Thus the end of new significant discoveries might not be very far away for copper and some of the other highly exploited rare metals.

Despite the fact that some new discoveries still are made, the sum of the reserves and the cumulative mine production of some metals, such as lead and nickel, has decreased over the last quarter of a century (Andersson and Råde 2001). Apparently, technological progress and new discoveries have not been able to compensate for increased costs due to (for example) environmental protection. In the years to come, technological progress will mitigate the escalation of recovery costs. Still, it might be overoptimistic to expect that the trends of cost reductions of the 20th century will continue in the 21st. Some have argued that implementation of environmental regulation worldwide and increased energy costs may increase recovery costs, and stronger forces working for the preservation of local environments may limit the access to land where ores are located (Hodges 1995). The preservation of land for tourism could, for example, prove more profitable than metal recovery. On the other hand, Tilton (1999) argues that real metal prices will not increase significantly over the next several decades, neither owing to ore depletion nor because of environmental regulation or land preservation. In our context we may note that, considering that the cost for nickel and lead already make up a relatively high share of the target cost for EV batteries (Table 31.1), the chance that EV manufactures could pay significantly higher prices for these metals appears to be limited.

Other metals that are mined (mostly) as main product or high-value co-product have reserve-to-extraction ratios that well exceed the 100 years envisioned as the time scale for the transformation of the energy system (Table 31.2). This is the case for platinum, lithium, rare earth elements and vanadium. For these metals annual availability may be a severer constraint than the available stock of resources. However, the mine production of these metals has grown by 2.5–5 per cent annually over the last three decades. Since they are main products, the mine production has the potential to grow at pace with the demand from an emerging technology. One obstacle may be that the mine production and the reserves are highly geographically concentrated: platinum and vanadium in South Africa, lithium in South America and rare earths in China. This increases the volatility of output and introduces uncertainty over the way economic development, monopolistic producer behavior, geopolitical relations and environmental considerations will affect the expansion of mine production.

The supply of metals that are mined as by-products will depend on the mine production of the main product. Tellurium and selenium are by-products of copper refining; indium, germanium and cadmium of zinc recovery, gallium of alumina production and ruthenium of the recovery of other platinum group metals. Gallium is about as abundant as copper and lead but seldom forms any mineral of its own (Figure 31.1). High enrichment is exceptional. This is also the case for the rarer element germanium and to some degree for selenium and indium. Tellurium and ruthenium exhibit higher enrichment factors but are less abundant. Cadmium is highly enriched in zinc ore but has a low economic value. Owing to the low grades in combination with relatively low prices, all these by-product metals typically generate less than 1 per cent of the revenues earned from the recovery of main products.

The by-product metals are not recovered from all mined ore where they are present and, where they are, recovery rates are often low. The recovery of cadmium can probably be improved only marginally, ruthenium by a factor of two and indium, tellurium and selenium possibly by a factor of two to six. The recovery of the more abundant metals, germanium and gallium, could possibly increase by up to two orders of magnitude. In addition recovery of metal by-flows mobilized by other large material flows, that is, waste mining (Ayres and Ayres 1996), could increase the annual availability. In the late 1990s, the germanium and gallium contents of combusted coals were about 500 times larger than the annual refinery production (Andersson 2000). As additional examples, cadmium could be recovered from phosphorous fertilizer, vanadium from petroleum refining and lithium from geothermal brines.

Table 31.3 *By-product values in zinc ore*

	Typical grade in zinc sulfide ores (ppm)	Price average 1993–7 (US$97/kg)	Metal value in ore (US$97/ton)
Zinc	40 000	1.1	44
Cadmium	200	2.3	0.4
Indium	4	280	1

Source: Andersson (2000).

Increased recovery would in most cases require higher metal prices. As is evident from Table 31.1 the PV metals share of the PV cost target is currently very low. If, for example, CIGS cells could produce 10 per cent more electricity per dollar than its competitors, PV manufacturers could pay up to 30 times the current indium price and still make a profit. This would imply an indium price not far from the gold price and indium would then become a high-value co-product in many zinc mines. At this price level indium demand could probably also affect mining rates. It might even make previously uneconomic deposits profitable (Table 31.3). As another example, cadmium availability constrains the expansion of nickel cadmium battery EVs, but the already considerable cost share of cadmium for EV batteries (Table 31.1) and its low value in zinc ore (Table 31.3) will inhibit cadmium from becoming a co-product of economic significance, at least owing to the demand from EVs.

If metal prices in general were to increase substantially, unconventional types of ore would become of economic interest. Manganese nodules on the ocean floor are believed to be a potential source of manganese, nickel, copper and cobalt, even though technical, economic and legal difficulties have pushed their exploitation into a distant future

(Hoagland 1993; Glasby 2000). These nodules could be profitable to mine if nickel or cobalt prices increased fourfold. It is questionable whether manufacturers of EV batteries could pay for such price increases. However, in some nodule samples, tellurium is reported to be present at high concentrations. A tellurium price level with gold and platinum prices could make nodules with tellurium grades of about 30 ppm economically recoverable for their tellurium, nickel, cobalt, copper and manganese contents (estimate based on Hoagland 1993). This is a speculative illustration of the important fact that technologies may be complementary at the material level. The growth or decline of one technology can affect the cost and availability of metal resources for another. (See, for example, Levine 1999 and Arthur 1996 for a discussion on complementarity or mutualism on the product use level. Economies of scope, in turn, would represent mutualism at the product manufacturing level.) As another example, the demand for platinum for fuel cells could generate by-product flows of ruthenium readily available for solar cells. To take a more imaginary example, demand for Li^6 for deuterium tritium fusion could pay for the recovery of very high-cost lithium resources and leave large amounts of Li^7 as a by-product available for EV batteries. By-product relations may stimulate a coevolution of technologies.

Secondary Resources and Competition for Metal

Mining per se does not decrease the stock of available resources. On the contrary, it makes the resources more available. It is the way that materials are used within society that can decrease the availability and turn resources into non-resources. For the 'mature' metal, lead, the cumulative mine production is four times larger than the reserves, implying that, if all of the cumulative past mine production of lead were readily accessible, economic lead resources would increase fivefold (Table 31.2). In contrast, the cumulative mine production of the 'younger' metals vanadium, lithium and platinum, adds only about 10 per cent to their respective reserves.

The cumulative production figures for the by-product metals (Table 31.2) denote refined metal. Given the low recovery factors for tellurium and indium, the amount contained in mined metal ores is estimated to be a factor of 10 larger (Andersson 2000). Of this, large amounts may still reside in tailings, slag and other wastes from primary extraction. As mentioned above, the mined amounts of gallium and germanium exceed the refined amounts by orders of magnitude.

From some end use products like fuel additives, paints and bullets, metals are dissipated quickly, even though a small fraction might be recovered by so-called 'phytomining' (Brooks *et al.* 1998). For metals in other products a controlled use is possible. Some of this metal has accumulated in landfills which possibly could be mined, while some is contained in long-lived products or circulated in recycling systems, such as lead in starter batteries and nickel in stainless steel.

For the future, continued economic growth is likely to create a fierce competition for some metals. The competitiveness of technologies varies. As indicated in Table 31.1, it might be difficult for EV batteries to compete for nickel on a price basis with current end uses such as stainless steel, whereas PV manufacturers probably could pay metal prices substantially above current levels. Of course, additional competition for metals from other new applications may well emerge. On the other hand, environmental regulations that restrict the use of toxic metals in applications where collection rates and recycling efficiency cannot reach close to 100 per cent could act in favor of toxic metal use in, for example, large batteries.

With a time scale of a century, it is not only the relative ability to pay among sectors that determines metal applications, but also *when* the demand arises. The diffusion of new technologies takes time, especially if new infrastructure is a precondition (Grübler and Nakicenovic 1991). To build up EV fleets or PV power parks of global scale will take several decades and a large proportion of the present reserves of metals with a relatively short extraction-to-reserve ratio, like lead, cadmium, zinc, copper and nickel, would by then have been mined and discarded or integrated in other technological systems. For indium, tellurium and selenium, that are by-products of zinc and copper with low recovery rates, this could also imply that large potential resources are lost in mining residues.

In summary, at present there are significant secondary resources of many by-product metals in residues from primary extraction. Scrapped goods cannot add more than marginally to the resources of most metals, with a few exceptions, most notably lead. More important for all metals is how residues from metal recovery, product manufacturing and used products are treated in the 50 years to come and whether metal could be transferred from one technological system to another at reasonable costs. It would be helpful, for instance, if EVs could inherit lead from starter batteries or platinum from car exhaust catalysts.

SEQUESTRATION, RUCKSACKS AND THE CADMIUM PARADOX: SOME ENVIRONMENTAL IMPLICATIONS

As pointed out by Lave *et al.* (1995), a large-scale introduction of technologies such as EVs that make use of toxic elements such as lead could increase the environmental loading of the metal. However, it is worth noting that this will only be the case if an increased mining of the metal is induced or if recovery and recycling are not more efficient than at present. As shown by Socolow and Thomas (1997) and Karlsson (1999), there is a potential that lead flows can be sufficiently closed within a battery production, use and recycling system. Large batteries and PV systems could be examples of applications where a controlled use of toxic metals is possible. They could then be used as a technospheric ‘attractor’ for toxic metals. A high demand for the metals would raise metal prices and discourage dissipative uses or even stimulate recovery of hazardous metals from various by-flows, such as cadmium from phosphate fertilizers, vanadium from petroleum refining, lithium from geothermal brines and a number of metals in ore tailings and coal ashes. This sequestration function of technologies has been noted by many, for example Ayres and Ayres (1996) and Kleijn (2000).

However, even if metals can be safely ‘stored’ in a technological system, one day the technology might be superseded by better performing alternatives. If no other large-scale applications exist that could inherit the metals, the scrap price would fall drastically and we would face the ‘riddance problem’: who would at that time pay for getting rid of a huge stock of hazardous waste in an environmentally acceptable fashion?

Higher metal demand could also lead to accelerated mining, which could increase sulfur dioxide emissions and metal leakage from waste dumps and cause large-scale land transformations. As shown above, this aspect may also be relevant for metals that currently are recovered as by-products. This gives rise to a 'cadmium paradox'. The presence of the toxic element cadmium is sometimes cited as a major problem for the use of alloys such as cadmium telluride (CdTe). But in line with the argument put forward above, 1GWp of CdTe PV modules could sequester 60Mg of cadmium that otherwise could have been wasted or used in short-lived products. On the other hand, if indium demand became a driver for the mining of zinc ore, 1GWp of CIGS modules could generate 3000–6000Mg of by-product cadmium. In this sense cadmium could become a larger problem for CIGS cells than for CdTe cells.

At present it seems unlikely that the by-product metals considered here would ever become main metals as a result of demand from the PV industry. But if it should happen, this industry could be held responsible for a large share of the environmental effects of mining. The potential PV has to dematerialize the energy system (Ogden and Williams 1989) would be turned into its opposite. As pointed out by Schmidt-Bleek (1994a), rare elements carry heavy 'rucksacks'. To recover 1g of tellurium from copper ore, one needs to mine more than 1t of ore. In the worst case, solar cells could become the 'coal industry' of the 21st century, extracting about as much material from the crust for every kWh of electricity produced.

SUSTAINABLE TECHNOLOGY TRAJECTORIES*

Are material constraints really a problem for technology evolution and economic growth? It has been argued that resource scarcity is not a threat to general economic growth in the long run (Barnett and Morse 1963; Smith 1979b). However, as shown above, specific designs of technologies are likely to be constrained by metal availability or associated environmental effects.

To manage a sustained industrialization and transform the energy system during this century, technologies such as PV and EVs must achieve high growth rates to gain from economies of scale and experience (Ayres and Martinàs 1992; Ayres and Axtell 1996). The question arises: what role may technologies which are promising in the short to medium term but material-constrained in the long-term play in such a development?

One crucial issue is the phenomenon of technology 'lock-in' due to the path-dependency of technological development (Arthur, 1994; David, 1985). This is a major theme in the growing literature on evolutionary economics, sociology of technology and history of technology; see, for example, Nelson and Winter (1982), Dosi *et al.* (1988), Freeman (1994), Bijker *et al.* (1987), Hughes (1983).

Here we suggest the following general terminology: there are *large-scale sustainable (LSS) technologies* and *constrained technologies*. The former are cornerstones of a sustainable industrial society. The latter are limited in some ways that disqualify them from being successfully implemented on the large scale for a long time. This constraint is likely to be physical or biological in nature or coupled to basic societal functions. If developed, the constrained technologies may play roles as *dead-end technologies* or *bridging technologie*; that is, they may defer or accelerate the development of technologies with greater

* For an extended discussion on the topic see Andersson (2001).

long-term potential, depending on how they mold the different systems the technologies are parts of.

Every technology is interwoven in various systems: (a) as a mix of materials it is a node in a material web; (b) as a technology, in the immaterial sense, it is an expression of accumulated knowledge; (c) as a product it is the result of production facilities and distribution networks; (d) as a service provider it is part of a use structure and the lifestyles of people; and (e) as a concept it is part of the world views and beliefs of people. The development of a dead-end technology would lead the technology evolution away from the development and diffusion of LSS technologies by creating material webs, skills, distribution channels and consumer behavior not compatible with LSS technologies, and thus make it more difficult to adopt these at a later stage. Bridging technologies would mold such systems in a direction compatible with LSS technologies and thus speed up the introduction of LSS technologies and function as a bridge on a sustainable technology path. Sometimes *transition technologies* filling a gap while waiting for LSS technologies are postulated. As technology development is path-dependent and does not occur in a vacuum, transition technologies other than as bridges or dead ends are very unlikely.

To give an example: PV and nuclear power plants could be perfect substitutes for an electricity consumer, but would involve very different regulations of operation, distribution channels, production facilities, knowledge bases and material webs, and as a consequence would not be easily substituted for each other. Thus an early market penetration of nuclear power is likely to 'lock out' PV. As another example, CdTe and amorphous silicon PV would be almost perfect substitutes for the operator and the electricity consumer; they could use the same distribution channels; they could in principle make use of some parts of the production line such as encapsulation and would partly build on the same knowledge base, but they would partly require different production facilities, knowledge bases and materials webs. While the diffusion of nuclear power is unlikely under any circumstances to function as a bridge to PV diffusion, the case of competing PV designs appears less clear. Whether CdTe and CIGS PV would function as a bridge to PV technologies with greater potential or as a dead end will depend on how policy and firm strategy are designed.

CONCLUSIONS

Of crucial importance in this analysis are the time scales and geographical scales involved. To manage a transformation of the energy system to cope with global warming and at the same time allowing for economic development of a growing world population will require major technical change during this century. Global implementation of new technologies takes many decades and thus technical change needs to continue at great speed in the right direction. Because of the need for scarce metals many technologies that appear promising in the short term are unlikely to be able to have an impact on the global scale which will be demanded in the longer term. However, these material constraints could be distant enough to allow for 'lock-in' of technology trajectories that are not sustainable. Large uncertainties exist regarding the requirement and availability of materials and what roles different technologies could play in the evolution of other technologies.

Implications for Policy and Strategy

To increase the potential of material-constrained technologies, the issue of metal intensity and utilization needs to be addressed in advance by manufacturers. Systems for closed loop recycling of new as well as old residues need to coevolve with the technology. Technology and policy that encourage waste mining and stockpiling of wastes with potential future value should be considered and thoroughly evaluated.

Long-term contracts or vertical integration between technology manufacturers, mining companies and recyclers could be a strategy to decrease the risk of sharply increased metal prices. The development of such strong alliances may, however, have an influence on the definition of environmental issues and could generate an undesirable technology 'lock-in'. Such a development needs therefore to be balanced with environmental and technology policy. To limit negative environmental effects attention needs to be paid to environmental issues related to the recovery of primary as well as the management of secondary materials all over the globe. To enable a rapid evolution along sustainable trajectories, a balance in public policy and firm strategy must be sought between the short-term requirement of cost reductions and the long-term requirement of keeping technological options open. If undesirable 'lock-in' can be avoided (or minimized) it is possible that material-constrained technologies could push the technology development in a desirable direction and function as bridges to large-scale sustainable technologies, and not as dead ends.

Implications for Industrial Ecology

Emerging energy technologies and related material flows appear to be a fruitful area of study for industrial ecology. Suitable tasks for industrial ecology could, for instance, be to study the recycling systems required for emerging technologies and assess potentials for waste mining. However, if industrial ecology is to be helpful in guiding the process of technology evolution, two key dimensions have to be acknowledged. As noted earlier, for example by Sagar and Frosch (1997) and Ruth (1998), industrial ecology needs to pay attention to the time dimension. We need to study the dynamics of the industrial ecosystems. How are efficient material webs designed and managed in a rapidly changing technology environment? How does drastically changing metal demand and prices affect primary extraction and recycling opportunities? How does the coevolution of technologies affect metal availability? How does the succession of technologies affect the management of technospheric metal pools?

Levine (1999) has suggested that industrial ecology should extend the scope from energy and material flows and the systems ecology model to products and the population ecology model. In practice, we presume this would imply the embracing of the whole area of evolutionary economics. As a more modest strategy, we suggest that industrial ecology could contribute with analyses of material systems in multidisciplinary studies of technology evolution, aiming at finding sustainable technology trajectories. Other subsystems would be studied in the context of evolutionary economics and sociology of technology.

A second key dimension is the geographical scale. Given the concentration of specific metal resources in a few locations on Earth and the worldwide scale of many technological systems, there are parts of the material webs for many elements that cannot be studied on the local scale. Industrial ecology thus needs to embrace the study of global scale material webs. The contrast between the dynamic global scale industrial ecology called for here and the prototype industrial ecology of Kalundborg (see Chapter 27) points to an interesting span within the field and to a challenge for the future.

32. Wastes as raw materials

David T. Allen

One of the central principles of industrial ecology is that industrial systems can develop the types of mass efficiency and cycling of materials exhibited by natural ecosystems. The vision shared by many in the industrial ecology community is of tightly integrated, mass-efficient manufacturing processes, which require energy inputs, but which require few mass inputs and generate little or no waste. Is such a vision realistic? Is it already beginning to occur? This chapter will examine the potential for using wastes as raw materials, by addressing each of the following questions: what are the flows and compositions of waste streams and how can their potential use as raw materials be assessed and promoted, and what types of design tools are necessary for increasing the use of wastes as raw materials?

FLOWS AND COMPOSITIONS OF WASTE STREAMS AND THE USE OF WASTES AS RAW MATERIALS

Advanced industrialized economies typically utilize 40–80 tons of material per year, per capita (Adriaanse *et al*. 1997). Comparative analyses of these material flows for the USA, Germany, Japan and Austria are provided elsewhere in this handbook; from these analyses it is clear that most materials used by highly industrialized economies are used once, then become wastes. Therefore most of the 40–80 tons of material used per year, per capita, become wastes. The sheer magnitude of waste generation is cause for concern and drives us to identify characteristics of the wastes, methods of waste management and the potential for reducing wastes.

Collecting information on the flow rates and compositions of waste streams is difficult. Data on waste streams and emissions are scarce, scattered and often inconsistent. Table 32.1 is a compilation of some of the data available on waste stream flows in the USA. Examination of this table indicates that some data are collected annually. Other data may be collected less than once a decade. Data are often reported separately for emissions to air, water and land. Data are also segregated by the characteristics of the wastes (for example, hazardous or non-hazardous). Confounding this scattering of information is the fact that units, reporting periods and the identification of the facilities reporting the wastes or emissions are generally not consistent between different sources of data. Nevertheless, with significant effort, diverse data on waste stream flow rates and emissions can be assembled into a comprehensive picture of material flows. Material flows in the USA will be used as an illustration.

The USA generates approximately 12 billion tons of waste per year, which is more than 40 tons per capita. The wastes can be in solid, liquid or gaseous states, and the majority

Table 32.1 A representative sampling of sources of data on industrial wastes and emissions in the USA

Non-hazardous Solid Waste
Report to Congress: Solid Waste Disposal in the United States, Vols I and II, US Environmental Protection Agency, EPA/530-SW-88-011 and EPA/530-SW-88-011B, 1988.

Criteria Air Pollutants
Aerometric Information Retrieval System (AIRS), US EPA Office of Air Quality Planning and Standards, Research Triangle Park, NC.
National Air Pollutant Emission Estimates, US EPA Office of Air Quality Planning and Standards, Research Triangle Park, NC.

Hazardous Waste (Air Releases, Wastewater and Solids)
Biennial Report System (BRS), available through TRK NET, Washington, DC.
National Biennial Report of Hazardous Waste Treatment, Storage, and Disposal Facilities Regulated under RCRA, US EPA Office of Solid Waste, Washington, DC.
National Survey of Hazardous Waste Generators and Treatment, Storage, Disposal and Recycling Facilities in 1986, available through National Technical Information Service (NTIS) as PB92-123025.
Generation and Management of Residual Materials; Petroleum Refining Performance (replaces the *Generation and Management of Wastes and Secondary Materials* series); American Petroleum Institute, Washington, DC.
Preventing Pollution in the Chemical Industry: Five Years of Progress (replaces the *CMA Hazardous Waste Survey* series), Chemical Manufacturers Association (CMA), Washington, DC.
Report to Congress on Special Wastes from Mineral Processing, US EPA Office of Solid Waste, Washington, DC.
Report to Congress: Management of Wastes from the Exploration, Development, and Production of Crude Oil, Natural Gas, and Geothermal Energy, Vol. 1, *Oil and Gas*, US EPA Office of Solid Waste, Washington, DC.
Toxic Chemical Release Inventory (TRI), available through National Library of Medicine, Bethesda, Maryland and RTK NET, Washington, DC.
Toxic Release Inventory: Public Data Release (replaces *Toxics in the Community: National and Local Perspectives)*; EPCRA hotline (800)-535-0202.
Permit Compliance System, US EPA Office of Water Enforcement and Permits, Washington, DC.

Economic Aspects of Pollution Abatement
Manufacturers' Pollution Abatement Capital Expenditures and Operating Costs, Department of Commerce, Bureau of the Census, Washington, DC.
Minerals Yearbook, Volume 1 Metals and Minerals, Department of the Interior, Bureau of Mines, Washington, DC.
Census Series: *Agriculture, Construction Industries, Manufacturers–Industry, Mineral Industries*, Department of Commerce, Bureau of the Census, Washington, DC.

Source: USDoE (1991).

of the waste flows are from industrial operations. Over the past decade, summaries of these material flows have been published by a number of authors (see, for example, Allen and Rosselot 1997 and references cited therein) and those summaries will not be repeated here. Instead, our focus will be on the type of information that is necessary to assess whether wastes really can be used as raw materials.

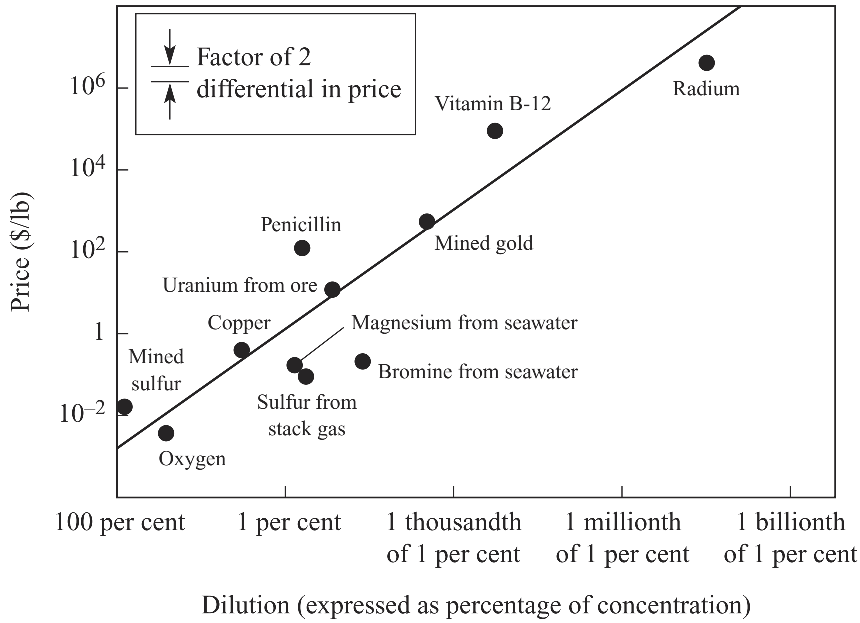
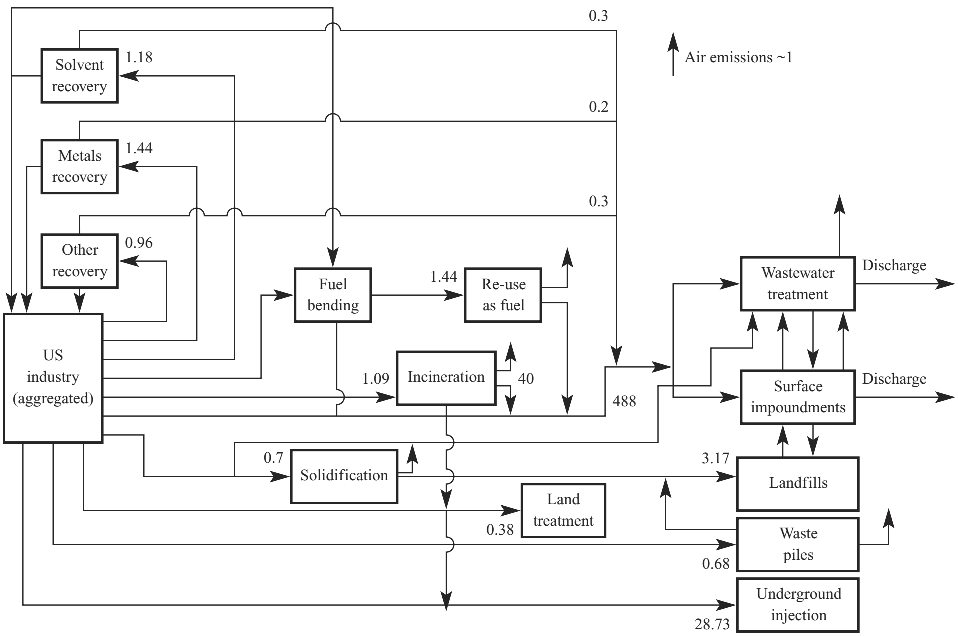


Figure 32.1 The Sherwood Plot

One of the most important pieces of information needed in assessing the potential for re-using materials in waste streams is the composition of the material. As shown in Figure 32.1, the value of a resource is proportional to the level of dilution at which valuable materials are present. The figure gives a 'Sherwood Plot', named after T. Sherwood of the Massachusetts Institute of Technology, who noted that selling prices of materials correlate with their degree of dilution in the initial matrix from which they are separated. Note that the horizontal axis shows an increasing degree of dilution, or decreasing concentration, in the initial matrix (US National Research Council 1987). Materials that are present at very low concentration can be recovered only at high cost, while materials present at high concentration can be recovered economically. Therefore, in evaluating whether wastes might be mined as raw materials, it is necessary to determine both the flow rate and the concentration of valuable materials in the waste. Unfortunately, relatively few of the data sources listed in Table 32.1 provide information about composition. One of the few sources of information on the concentrations of materials in wastes, in the USA, is the National Hazardous Waste Survey (described at length in Allen and Jain 1992). This database combines detailed data on waste composition with information on bulk waste stream properties. The database includes all waste streams regulated as hazardous wastes under the Resource Conservation and Recovery Act. The total mass flow rate of all materials represented in the National Hazardous Waste Survey is approximately 0.75 billion tons per year, and therefore the data represent only 5–10 per cent of the total flow rate of industrial wastes. So it is far from comprehensive and the data are becoming dated, but it still represents some of the best information available in the USA on waste composition.

A summary of the hazardous waste flows reported in the database is provided in Figure



Source: Allen and Jain (1992).

Figure 32.2 Flow of industrial hazardous waste in treatment operations (1986 data in millions of tons per years)

32.2 (Allen and Jain 1992). The data are for 1986, the only year for which survey data are available. As indicated in the figure, a small fraction of solvent, metal and other wastes, less than 1 per cent of total waste mass generated, flows through recycling loops. The total mass involved in recycling is about 5 million tons per year (Mt/yr). The largest single category of streams in terms of mass flow, nearly 720Mt/yr (more than 90 per cent of the total waste flow represented in the database), is hazardous wastewater. Most of this stream is water with a small percentage of non-aqueous contaminants; hence the mass of the chemically hazardous component of this stream is within an order of magnitude of the components being recycled. A third set of waste streams, about 4Mt/yr, is sent to various thermal treatment technologies which include direct incineration, fuel blending and re-use as fuel.

While an examination of total waste flows is a necessary first step in assessing the use of waste streams as raw materials, total mass is not a good indicator of the potential value of waste streams. Instead, the concentration and mass flow rates of valuable resources in the waste streams will be the most important evaluation criteria.

As a simple case study of evaluating the potential use of industrial wastes as raw materials, consider the flows of metals in hazardous waste streams. The Sherwood diagram (Figure 32.1) showed that, whereas materials such as gold and radium can be recovered from raw materials that are quite dilute in the resource, materials such as copper can be recovered economically only from relatively rich ores. The price of a metal can therefore be used to estimate the approximate concentration at which the metal can be recovered from a hazardous waste stream. The approximate concentration at which metals can be effectively recovered from wastes can then be compared to the actual concentrations of the materials in the wastes. To illustrate this evaluation of waste stream composition data, Figures 32.3 and 32.4 show the concentrations at which copper and zinc are found in hazardous waste streams, as documented in the National Hazardous Waste Survey. Surprisingly, many of the waste streams contain relatively high concentrations of these metals. Approximately 90 per cent of the copper and 95 per cent of the zinc found in hazardous wastes is at a concentration high enough to recover. Table 32.2 shows that the situation for copper and zinc is not unusual. In fact, for every metal for which data existed in the National Hazardous Waste Survey, recovery occurred at rates well below rates that would be expected to be economically viable.

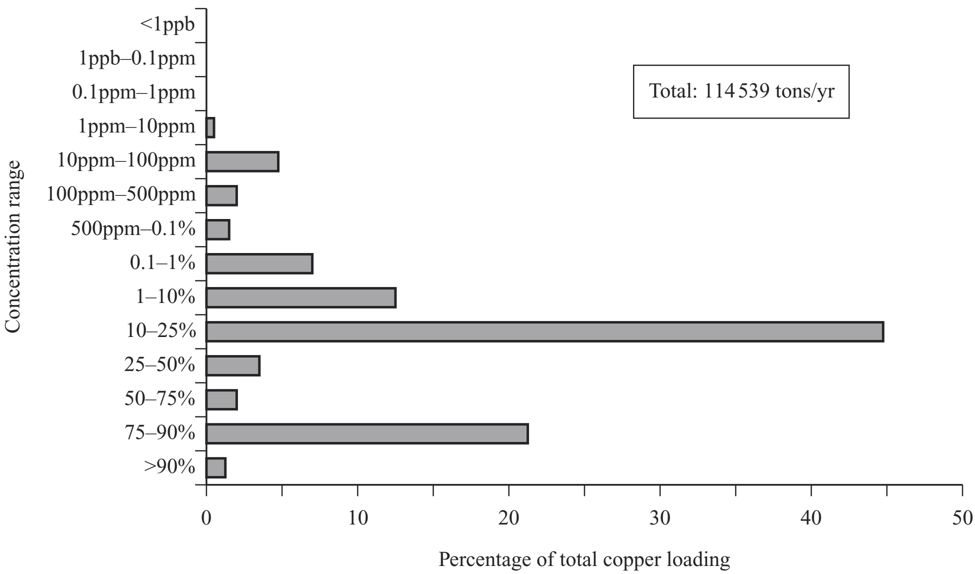


Figure 32.3 Concentration distribution of copper in industrial hazardous waste streams

This very focused analysis, initially performed in 1994 (Allen and Behmanesh 1994) led us to conclude that many opportunities existed for recovering materials from wastes. There are limitations to the analysis, however. The analysis focused only on hazardous wastes, where liability concerns may limit the desire to recycle. The identification of 'recyclable' streams was simplistic. It ignored issues related to economies of scale (that is, processing geographically dispersed, heterogeneous waste streams may be more expensive than extracting a relatively homogeneous ore from a single mine). Nevertheless, the analysis indicated that resources are not effectively recovered from many waste streams. One of the primary barriers to using wastes as raw materials is a lack of critical information on waste streams. While a large number of data sources are available on waste streams, they lack critical information that is needed to assess whether waste streams might be re-used. Data on the composition of wastes, their location and co-contaminants are rarely available, yet are critical to evaluating the potential use of wastes as raw materials. Lack

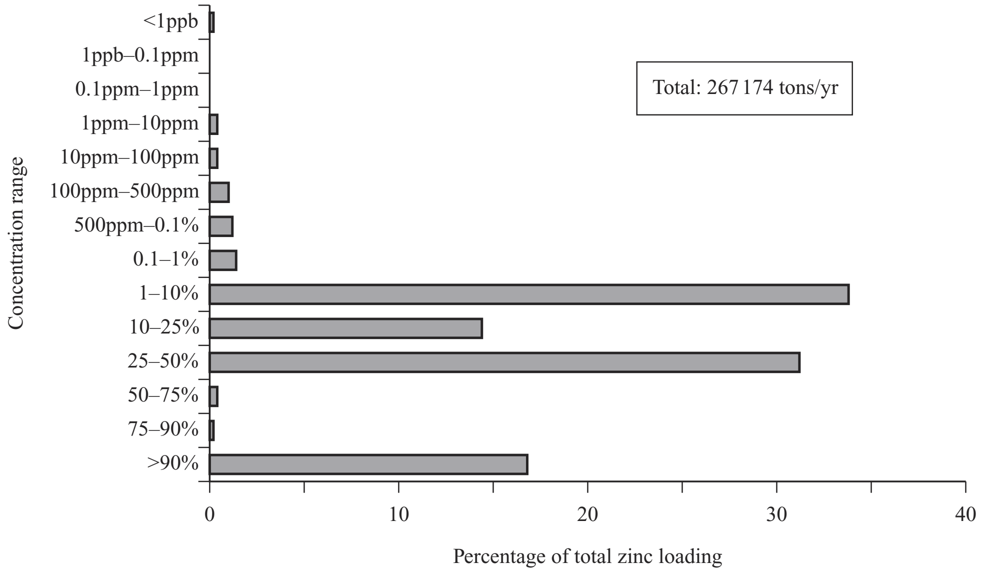


Figure 32.4 Concentration distribution of zinc in industrial hazardous waste streams

Table 32.2 Percentage of metals in hazardous wastes that can be recovered economically

Metal	Theoretically recoverable (%)	Recycled in 1986 (%)
Sb	74–87	32
As	98–99	3
Ba	95–98	4
Be	54–84	31
Cd	82–97	7
Cr	68–89	8
Cu	85–92	10
Pb	84–95	56
Hg	99	41
Ni	100	0.1
Se	93–95	16
Ag	99–100	1
Tl	97–99	1
V	74–98	1
Zn	96–98	13

Source: as estimated by Allen and Behmanesh (1994).

of data is not the only barrier to using wastes as raw materials, however. Three additional reasons for the failure of resources recovery (adapted from Allen 1993) are (a) lack of a recycling infrastructure, (b) regulatory barriers, and (c) technological limitations.

Lack of a Recycling Infrastructure

Potentially valuable materials may be discarded as wastes simply because of the lack of necessary infrastructure. Lead and nickel recovery provide interesting illustrations of this point. Lead is efficiently recycled because a collection network is available which will transport lead-containing wastes, such as used batteries, to a processing facility able to take advantage of economies of scale. As a consequence, over 90 per cent of all car batteries are recycled and more than 50 per cent of all lead in hazardous waste is recovered. In contrast, the recycling infrastructure for nickel is far less developed. Nickel is extensively used in batteries and the technology for recovering nickel from these batteries exists (see, for example, Steele and Allen 1998), but the batteries are not recycled because a material collection infrastructure and the necessary processing capacity has not developed. As suggested by Table 32.2, the lack of a recycling infrastructure also inhibits the recycling of nickel in industrial wastes.

Regulatory Barriers

One of the primary barriers to material re-use and resource recovery in the USA is the complex set of rules and definitions that determine how a waste is to be managed. A case study of silver recycling in the San Francisco Bay area provides an interesting case study of the role that regulatory barriers can play in limiting resource recovery (Kimbrough *et al*. 1995).

Because of its ecotoxicity, very strict discharge limits are placed on silver in the San Francisco Bay area. Yet, despite these strict limits, silver discharges were still a problem, and the source of the problem did not become clear until the material flows – the industrial ecology – of silver were examined. In the USA, approximately 4000 tons of silver are consumed annually. Slightly over 50 per cent of this usage is in photographic and radiographic materials, and a large fraction of this silver becomes part of a waste stream, typically as silver halides in developer solutions. The silver halides in developer solutions are typically at concentrations high enough to be regulated and high enough to be economically recycled, yet a large fraction of developer solution is not recycled because it is generated, in quantities of 5 gallons per month or less, by small waste generators (primarily medical and dental offices). In California these small generators could economically recycle the silver in their developing solution. However, to do so would have required paying a regulatory fee of around \$125 to pick up the spent developer. If assaying and refining fees were added to the hauling cost, the total transaction cost became about \$200. So a silver waste generator had to process over \$200 worth of silver to make recycling economical. This would mean processing approximately 250 gallons of developer with a silver loading of 1 gram per liter. Because most facilities generated only about 5 gallons of developer per month, storing enough solution for economical recycling was impractical. A variety of on-site silver recovery technologies are available and could be alternatives to hauling silver off-site for recovery. Until recently, however, small generators in

California that treated such waste on site had to obtain permits that cost approximately \$1000 for the first year and \$500 each year thereafter. This meant that the generator had to recover the silver present in approximately 1250 gallons of developer fluid just to recoup the first year's permitting fee.

In this example, regulatory fees presented a significant barrier to silver recovery for medical and dental offices, but innovative technologies and restructuring the regulatory approach for these generators can and did result in a significant increase in silver recovery.

Technological Limitations

In addition to regulatory barriers to technology development, there may be some technological limits to resource recovery. Although a comprehensive study of the issue has yet to be performed, some observations about current practices may be illuminating. Consider Figure 32.2, which provides an overview of hazardous waste management practices in the USA. A striking feature of waste management practices is the relatively limited range of technologies employed. Combustion, land disposal, solvent recovery, metal recovery and a few types of wastewater treatment technologies dominate. Notably absent is the impact of waste exchanges, catalytic reduction and other resource recovery methods. A quantitative analysis of the technological potential for resource recovery may remain elusive, but the Gibbs–Dühem equation of thermodynamics can provide an upper bound on the range of possibility. A simple calculation reveals that the entropy that needs to be overcome in separating a pound of pure material from a million pounds of waste has an energy equivalent of a few hundred to a thousand BTU (the amount of energy in about an ounce of gasoline). Although this calculation is highly idealized, it reveals that fundamental physical laws do not inhibit resource recovery from dilute streams. Rather, the limits are technological.

Summary

Many industrial waste streams contain valuable resources and are suitable for use as raw materials. Waste streams from a range of industrial sectors must be compared to the raw materials and process streams used in other sectors to identify opportunities for re-use. These material re-use opportunities can be identified using the tools of systems engineering. Some of the resulting industrial networks may emerge as Ecoparks, with geographically co-located facilities, as described elsewhere in this handbook. Other industrial networks that arise may exchange materials over great distances, still preserving mass efficiency. No matter what the structure, however, design tools will be necessary to identify the networks. The next section describes such design tools, focused on the chemical manufacturing sector.

DESIGN TOOLS FOR PROMOTING THE USE OF WASTES AS RAW MATERIALS

This section presents two examples of the types of engineering design tools that will need to develop if wastes are to be systematically used as raw materials. The first example deals with water re-use in chemical manufacturing. Water re-use was chosen as a case study for material re-use because water is becoming an increasingly scarce and valuable resource worldwide, and because water provides a rich modeling opportunity: the number of water and wastewater streams in any group of industries is high and the set of possibilities for re-use is large. The second example of material re-use design will examine chlorine use in chemical manufacturing. In this case, the chemistry involved in the analysis is much more complex, but the underlying design tools remain the same.

Water Re-use in Chemical Manufacturing

Rising water costs, limited water supplies, waste minimization and pollution control issues are compelling industrial users of water to consider water reclamation, re-use and recycling. Currently, most wastewater is treated and released into receiving waters. However, in many cases it is feasible for treated wastewater to be re-used because certain water uses (for example, irrigation, manufacturing and sanitation) do not require the high-quality water they now receive. If wastewater is re-used, then total water demand and effluent treatment load can be lowered.

Despite their potential, water reclamation, re-use and recycling technologies remain greatly underused. Furthermore, most industrial water re-use focuses on recycling and process modifications within one facility. Very little attention has been given to the possibility of water exchange among industries, even though integrated water re-use management has been recommended as an effective means of water conservation. Integrating water re-use throughout a region, rather than merely within a single facility, provides economies of scale and more re-use opportunities. Regional integration also provides a systematic framework in which to overcome the legal and public perception impediments to water re-use. However, previous regional reclamation projects have faced difficulty in identifying users for reclaimed water. These shortfalls have been attributed to insufficient planning and design.

Planning and design of water re-use programs at a regional level require not only traditional information about the quantity and quality of water supply and demand, but also information about the geographical location where the supply and demand occur. Traditional approaches to water re-use have not included explicit quantitative geographical data, even though conveyance and distribution systems make up the principal costs of water exchange projects, and these costs depend primarily on geographic considerations such as distance between distributor and receiver, and elevation differences (for pumping). Therefore, in evaluating water re-use (and re-use of other materials), geographical information becomes critical. Geographical Information Systems (GIS) can be used as a tool to incorporate spatial information into a water re-use and other material re-use analyses.

GIS is an organized collection of computer hardware, software, geographic data and personnel designed to efficiently capture, store, update, manipulate, analyze and display all forms of geographically referenced information. GIS integrates database operations with the unique visualization benefits and geographic analysis offered by maps. Thus GIS is an excellent framework in which to combine industry water characteristics and geographic planning considerations such as pipeline locations for transporting water. Nobel and Allen (2000) applied a GIS framework to identify water re-use opportunities in the

Bayport Industrial Complex in Pasadena, Texas. For approximately 20 industrial facilities, opportunities for re-using water were evaluated, by comparing water discharge characteristics and water inlet quantity and quality requirements. The distances and elevation changes between facilities were evaluated using GIS. When these geographical data were combined with the water quality data, a series of mathematical constraints on water re-use resulted. These were then solved using mathematical programming techniques, and the results were illustrated in maps. Figure 32.5 is one of the maps. In this scenario, water reclaimed at the wastewater treatment plant (WWTP) and fresh water processed at the water treatment plant are assumed to have equal costs at equal levels of water quality. The distribution network in this scenario is based on distance from the processing facility. Arrows indicate water pathways and the thickness of the arrows is an indication of flow rate. Flow rates are given in 1000 gallons per day. Other scenarios are reported by Nobel and Allen (2000).

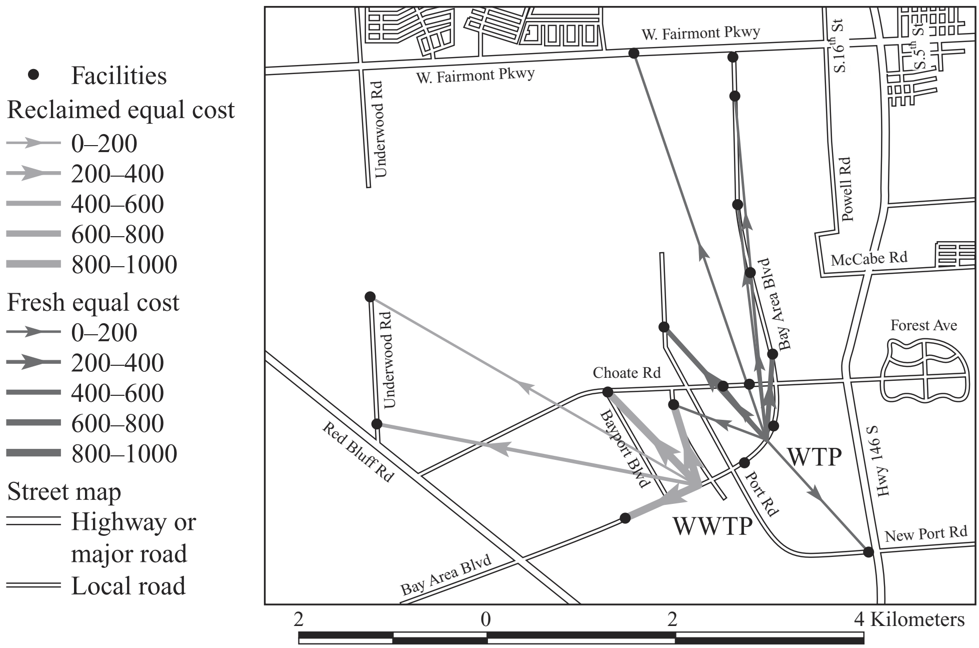


Figure 32.5 Optimal supply network for waste re-use in the Bayport Industrial Complex

This case study, which is described in more detail elsewhere (Keckler and Allen 1998; Nobel and Allen 2000), illustrates the importance of using geographical information in evaluating the re-use of materials. This is particularly important for a material such as water, where transportation costs can be significant relative to material costs. This example is simplistic, however, in its treatment of the chemistry frequently involved in the re-use of materials. The following example illustrates some of the complexities that can arise in re-using materials that are employed for their chemical rather than their physical properties (as is the case for water).

Chlorine Re-use in Chemical Manufacturing

The environmental performance of chemical processes is governed not only by the design of the process, but also by the way the process integrates with other processes and material flows. Consider a classic example – the manufacture of vinyl chloride.

Billions of pounds of vinyl chloride are produced annually. Approximately half of this production occurs through the direct chlorination of ethylene. Ethylene reacts with molecular chlorine to produce ethylene dichloride (EDC). The EDC is then pyrolyzed, producing vinyl chloride and hydrochloric acid.

$$Cl_2 + H_2C = CH_2 \rightarrow Cl\ H_2C - CH_2\ Cl.$$

$$Cl\ H_2C - CH_2\ Cl \rightarrow H_2C = CH\ Cl + HCl.$$

In this synthesis route, one mole of hydrochloric acid is produced for every mole of vinyl chloride. Considered in isolation, this process might be considered wasteful. Half of the original chlorine winds up, not in the desired product, but in a waste acid. But the process is not operated in isolation. The waste hydrochloric acid from the direct chlorination of ethylene can be used as a raw material in the oxychlorination of ethylene. In this process, hydrochloric acid, ethylene and oxygen are used to manufacture vinyl chloride.

$$HCl + H_2C = CH_2 + 0.5O_2 \rightarrow H_2C = CHCl + H_2O.$$

By operating both the oxychlorination pathway and the direct chlorination pathway, the waste hydrochloric acid can be used as a raw material and essentially all of the molecular chlorine which originally reacted with ethylene is incorporated into vinyl chloride. As shown in Figure 32.6, the two processes operate synergistically and an efficient design for

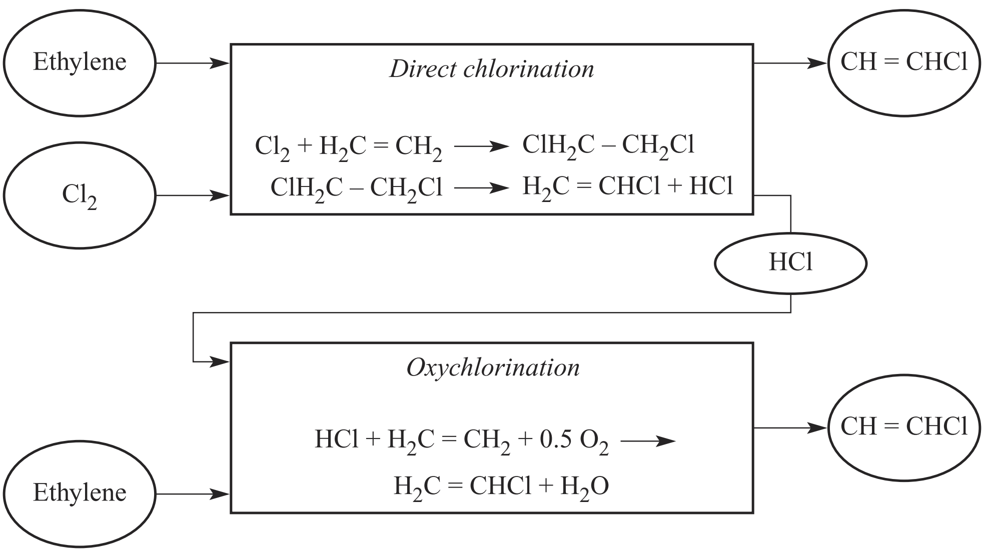


Figure 32.6 Direct chlorination and oxychlorination of ethylene in tandem

the manufacture of vinyl chloride involves both processes. By-product hydrochloric acid from the direct chlorination of ethylene is used as a raw material in the oxychlorination process; by operating the two processes in tandem, chlorine is used efficiently.

Additional efficiencies in the use of chlorine can be obtained by expanding the number of processes included in the network. In the network involving direct chlorination and oxychlorination processes, both processes incorporate chlorine into the final product. Recently, more extensive chlorine networks have emerged, linking several isocyanate producers into vinyl chloride-manufacturing networks (McCoy 1998). In isocyanate manufacturing, molecular chlorine is reacted with carbon monoxide to produce phosgene:

$$CO + Cl_2 \rightarrow COCl_2.$$

The phosgene is then reacted with an amine to produce an isocyanate and by-product hydrochloric acid:

$$RNH_2 + COCl_2 \rightarrow RNCO + 2\ HCl.$$

The isocyanate is subsequently used in urethane production, and the hydrochloric acid is recycled. The key feature of the isocyanate process chemistry is that chlorine does not appear in the final product. Thus chlorine can be processed through the system without being consumed. It may be transformed from molecular chlorine to hydrochloric acid, but the chlorine is still available for incorporation into final products, such as vinyl chloride, that contain chlorine. A chlorine–hydrogen chloride network incorporating both isocyanate and vinyl chloride has developed in the Gulf Coast of the USA. The network is shown in Figure 32.7. Molecular chlorine is manufactured by Pioneer and Vulcan Mitsui. The molecular chlorine is sent to both direct chlorination processes and to isocyanate manufacturing. The by-product hydrochloric acid is sent to oxychlorination processes or calcium chloride manufacturing. The network has redundancy in chlorine flows, such that most processes could rely on either molecular chlorine or hydrogen chloride.

Consider the advantages of this network to the various companies (C.G. Francis, personal communication 2000):

- Vulcan/Mitsui effectively rents chlorine to BASF and Rubicon for their isocyanate manufacturing; the chlorine is then returned in the form of hydrochloric acid for ethylene dichloride/vinyl chloride manufacturing.
- BASF and Rubicon have guaranteed supplies of chlorine and guaranteed markets for their by-product HCl.

Even more complex networks could, in principle, be constructed. As shown in Table 32.3, chlorine is used in manufacturing a number of non-chlorinated products. The table lists, for selected reaction pathways, the pounds of chlorinated intermediates used along the supply chain, per pound of finished product. This ranking provides one indication of the potential for networking these processes with processes for manufacturing chlorinated products (see Rudd *et al.* 1981; Chang and Allen 1997).

An examination of individual processes, such as those listed in Table 32.3, can be useful in building process networks, but the individual process data do not reveal whether effi-

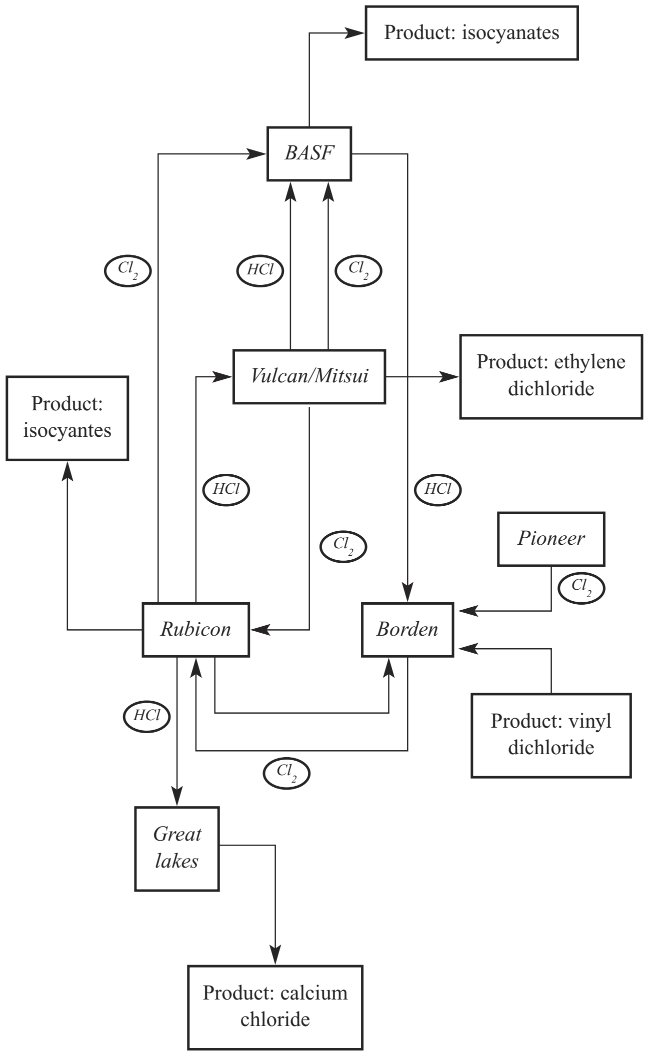


Figure 32.7 Chlorine flows in combined vinyl chloride and isocyanate manufacturing

cient use of chlorine is a major or a minor issue in chemical manufacturing. To determine the overall importance of these flows, it is useful to consider an overall chlorine balance for the chemical industry. The overall flows of chlorine into products and wastes, as well as the recycling of chlorine in the chemical manufacturing sector, are shown in Figure 32.8. The data indicate that roughly a third of the total chlorine eventually winds up in wastes. By employing the types of networks shown in Figures 32.6 and 32.7, the total consumption of chlorine could be reduced.

Identifying complex material re-use routes, such as those used for chlorine, is difficult and needs to rely on comprehensive, integrated models of material flows in the chemical process industries. Fortunately, such models have been developed. Rudd and co-workers

Table 32.3 Partial listing of non-chlorinated chemical products that utilize chlorine in their manufacturing processes

Product	Synthesis pathway	Pounds of chlorinated intermediates per pound of product
Glycerine	Hydrolysis of epichlorohydrin	4.3
Epoxy resin	Epichlorohydrin via chlorohydrination of allyl chloride, followed by reaction of epichlorohydrin with bisphenol-A	2.3
Toluene diisocyanate	Phosgene reaction with toluenediamine	2.2
Aniline	Chlorobenzene via chlorination of benzene, followed by reaction of chlorobenzene with ammonia	2.2
Phenol	Chlorobenzene via chlorination of benzene, followed by dehydrochlorination of chlorobenzene	2.1
Methylene diphenylene diisocyanate	Phosgene reaction with aniline (also produced with chlorinated intermediates)	1.5
Propylene oxide	Chlorohydration of propylene	1.46

Source: Chang and Allen (1997).

have developed basic material and energy flow models of over 400 chemical processes associated with the production of more than 200 chemical products (Rudd *et al.* 1981), describing a complex web of chemical manufacturing technologies.

An understanding of material flows in these networks can be used at a variety of levels. First, the material flow networks can be used simply to identify potential users and suppliers of materials, and to identify networks of processes that are strategically related. For example, for the types of networks shown in Figures 32.6 and 32.7, it would be useful to have lists of processes that produce and consume hydrochloric acid. A partial list is given in Table 32.4; such lists are useful in identifying potential material exchange network.

Once consumers and producers of the target chemicals are identified, material and energy flow models can be used to construct networks. The network that makes the most sense depends on the features that are to be optimized. Analyses have been performed to identify networks that minimize energy consumption (Sokic, Cvetkovic and Trifunovic 1990; Sokic, Zdravkovic and Trifunovic 1990), the use of toxic intermediates (Yang 1984; Fathi-Afshar and Yang 1985) and chlorine use (Chang and Allen 1997). Other analyses have considered the response of networks to perturbations in energy supplies (Fathi-Afshar *et al.* 1981) and restrictions on the use of toxic substances (Fathi-Afshar and Rudd 1981). Regardless of the application, however, the material flow model of the chemical manufacturing web provides the basic information necessary to identify and optimize networks of processes.

Yet another use of comprehensive material flow models is in the evaluation of new technologies (Chang and Allen 1997). Consider once again the case of chlorine use in chem-

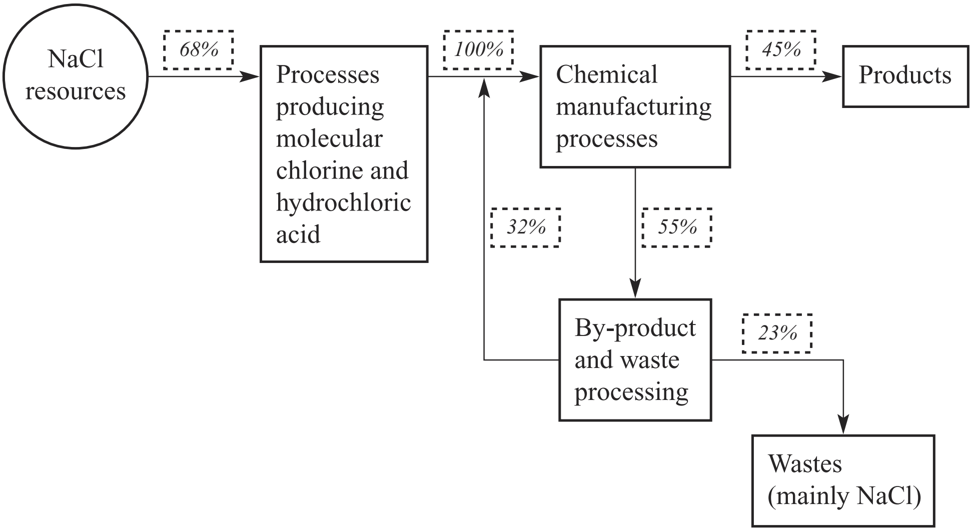


Figure 32.8 A summary of chlorine flows in the European chemical industry

Table 32.4 Partial list of processes that produce or consume hydrochloric acid

Processes that consume hydrochloric acid	Processes that produce hydrochloric acid
Chlorobenzene via oxychlorination of benzene	Adiponitrile via chlorination of butadiene
Chloroprene via dimerization of acetylene	Benzoic acid via chlorination of toluene
Ethyl chloride via hydrochlorination of ethanol	Carbon tetrachloride via chlorination of methane
Glycerine via hydrolysis of epichlorohydrin	Chloroform via chlorination of methyl chloride
Methyl chloride via hydrochlorination of methanol	Ethyl chloride via chlorination of ethanol
Perchloroethylene via oxychlorination of ethylene dichloride	Methyl chloride via chlorination of methane
Trichloroethylene via oxychlorination of ethylene dichloride	Perchloroethylene via chlorination of ethylene dichloride
	Phenol via dehydrochlorination of chlorobenzene
	Trichloroethylene via chlorination of ethylene dichloride

Note: *Such lists are useful in identifying potential material exchange networks.

ical manufacturing. Rather than generating complex networks involving HCl and molecular chlorine, it might be preferable to use a chemistry that converts waste HCl into molecular chlorine. Several processes have been proposed and three are listed in Table 32.5. These processes will only be successful if they can compete with the re-use of by-product HCl, in the types of networks described in Figures 32.6 and 32.7. Data on material and

Table 32.5 Processes for reducing chlorine use in chemical manufacturing

Process description
Chlorine via electrolysis of hydrogen chloride (Ker-Chlor process)
Chlorine via oxidation of hydrogen chloride ($CuCl_2$ catalyst
Chlorine via oxidation of hydrogen chloride (HNO_3 catalyst)

Source: Chang and Allen (1997).

energy flows in the chemical manufacturing web can again be used to assess the competitiveness of new chemical pathways, such as the technologies listed in Table 32.5.

Summary

This section has emphasized that the environmental performance of chemical processes is governed not only by the design of the process, but also by the way the process integrates with other processes. Integration with other processes can occur through exchanges of material, through exchanges of energy and through common use of utilities, such as cooling and process waters. To design efficient and economical processes, designers must systematically search out markets for by-products; they should consider using by-products from other processes as raw materials; and, perhaps most significantly, they should not restrict their searches to chemical manufacturing processes.

CONCLUSION

At the beginning of this chapter, two basic questions were posed: what are the flows and composition of waste streams and how can their potential use as raw materials be assessed and promoted, and what types of design tools are necessary for increasing the use of wastes as raw materials? The case studies and data presented in this chapter have demonstrated that waste flows are substantial and that many wastes have the potential to be used as raw materials, creating an industrial ecology. The design tools that will enable the creation of these industrial ecosystems are just beginning to emerge and will need to involve a sophisticated understanding of the composition and chemistry of wastes as well as an understanding of the role of spatial data.

33. Heavy metals in agrosystems

Simon W. Moolenaar*

Agrosystems belong to the biosphere as well as the anthroposphere. They serve not only as significant sources of energy and matter, but also as sinks for many residual fluxes. Soils are vital constituents of agrosystems. Important functions include habitat protection for flora and fauna, contribution to global nutrient cycling, the bearing function, the filtering or buffer function, and others. Soil quality especially influences the quality of groundwater, which may serve as a resource for drinking water or as surface water recharge (de Haan 1996; Blum 1990; Harris *et al* 1996). One aspect of soil quality is the accumulation of heavy metals in soil. An analysis of the input and output fluxes of Cd, Cu, Pb and Zn in agriculture and of their resulting accumulation in agricultural soils is necessary to ensure sustainable management of these metals in agricultural systems.

Agrosystems may be viewed as 'domesticated ecosystems' intermediate between natural ecosystems (such as wild forest) and fabricated systems (for example, a city). Both agrosystems and natural ecosystems are solar-powered and are composed of primary producers, consumers and decomposers. Agrosystems differ from natural systems in that they use manufactured products, including synthetic fertilizers, pesticides and hybrid seeds, as well as fossil fuels, while species diversity is greatly reduced by human management to optimize yields of desired products. The dominant plants and animals are under artificial rather than natural selection, and control is external to the system rather than via feedback and self-regulation as in natural ecosystems.

Concern about the entry of various heavy metals into the environment and hence into the human food chain was first expressed by Nriagu in several classic papers (1979, 1980a, 1980b, 1980c, 1988, 1989, 1990; Nriagu and Pacyna 1988; Nriagu and Davidson 1980). He described the long-term exposure of mankind to elevated environmental levels of metals as an 'experiment in which one billion human guinea pigs are being exposed to undue insults of toxic metals' (Nriagu 1988, p.139). The natural biogeochemical cycle of heavy metals mainly consists of wind-borne soil particles, sea-salt spray, volcanoes, forest fires and other biogenic sources. The main anthropogenic sources of heavy-metal emission are mining and smelting of metal ores, incineration of wastes, car exhausts and fossil fuel combustion. More recent studies of the flows of toxic metals in the environment include Ayres and Rod (1986), Nriagu and Pacyna (1988), Alloway (1990, 1995), Baccini and Brunner (1991), Stigliani *et al.* (1993), Ayres and Ayres (1994), Moolenaar (1998), Guinée *et al.* (1999) and van der Voet *et al.* (2000a).

Transfer to the human food chain is an important environmental issue because toxic metals are non-degradable (hence persistent) and the continuing build-up of such toxins in life-support systems entails (partly unknown) ecosystem and health risks. For these

* This chapter is based in part on Moolenaar (1999) and Moolenaar and Lexmond (1999, 2000).

reasons a number of policy measures have been drafted in both national and international forums to reduce potential ecological and human health risks caused by excessive concentrations of heavy metals in environmental media and agricultural produce. Such measures have been partially successful with respect to a reduction of industrial emissions to water and air (for example, VROM 1993). However, overall rates of primary production and consumption of most metals have not decreased, but rather have increased. This implies a continuing accumulation in capital goods, intermediate products and/or in recycling processes. Consumption-related emissions to air, water and soil may well increase in the future as a result of this accumulation (Guinée *et al.* 1999).

The buffering capacity of soil may be defined as the capacity of soil to inactivate contaminants. Inactivation of heavy metals is mainly achieved by effective bonding of the metal onto soil constituents, such as clay, or conversion into insoluble compounds. Soil is considered to be polluted when the buffering capacity is exceeded. Buffering capacity – hence vulnerability – varies widely for different compounds and different soils. Because of buffering capacity, it usually takes some time before negative effects of a toxic contaminant's presence become apparent. Pollution of the soil is indicated by an excessive presence or availability of compounds (de Haan 1996). For instance, both essential metals (Cu and Zn) and non-essential metals (Cd and Pb) become toxic when critical levels are exceeded, and adversely affect biodiversity, soil productivity and overall functioning of agrosystems (Ross 1994).

SCALES OF ANALYSIS

Substance flow analysis on the national scale may be carried out by calculating the national balance: subtracting all the heavy-metal flows leaving agriculture from all the heavy-metal flows entering agriculture. Thus the total net input of heavy metals to the (agricultural) soil gives an overview of the 'average burden' on 'average agricultural soil' by applying national statistics on feedstuffs, mineral fertilizers, animal manure, agricultural products (milk, meat, crops) and so on. Using averages of annual sales, compositions (for example, of crops or fertilizers), application rates and yields per crop, the mean annual loads in a certain region can be calculated. The information on composition may be obtained by direct measurements or by using statistical and bookkeeping data, or (usually) a combination of the two. In either case the numbers give a 'snapshot' at a certain point in time. Regular monitoring is the only way to check on variability and time trends.

Dutch, Danish and Finnish studies illustrate the large uncertainties in quantifying metal flows on the national scale. Inputs of heavy metals by (transboundary) air pollution and imports of feed concentrates (containing Cd, Cu, Zn) and phosphate rock (containing Cd) are striking (Poppe *et al* 1994; de Boo 1995). In Denmark, the effects of legislation on decreasing the Cd surplus on the balance have been quite positive. The prohibition of cadmium in pigments and other products, combined with stack gas cleaning, has greatly reduced atmospheric emissions in the last few years. Moreover, the Cd contents in phosphate fertilizers have also decreased (Hovmand 1984; Tjell and Christensen 1992). In Finland, the low deposition rate of Cd and the exceptionally low Cd input with phosphate fertilizers has (on average) led to a steady-state situation (that is, no net accumulation) for Cd in arable soils. However, copper and zinc added in large amounts to feeds result in large Cu and Zn flows in animal manure (Mäkela-Kurtto 1996). Average numbers on net accumulation of heavy metals in soil are given in Table 33.1.

Table 33.1 Net heavy-metal accumulation for some European soils (grams per hectare per year)

	Netherlands	Denmark	Finland
Cd	1.6	1.5	0.21
Cu	340		153
Pb	17		5.5
Zn	800		268

Sources: The Netherlands, de Boo (1995); Denmark, Tjell and Christensen (1992); Finland, Mäkela-Kurtto (1996).

The farm-gate balance quantifies the characteristic input and output flows at the farm level. Analysis on the farm scale does not distinguish the soil, animal and plant compartments within the farm boundaries. The flows between the farm compartments (that is, internal flows) are not accounted for in a farm-gate balance although these internal flows may play an important role, as is the case in dairy and mixed farming. Analysis is sometimes also carried out on the individual field scale. This relates the rates of metal accumulation, inputs and outputs with regard to the soil compartment; that is, the plow layer (about 0.3 m) of individual fields.

The scale on which processes are studied may influence the conclusions. Different scales of analyses may reveal different problem flows. Farm-gate balances show the total contaminating potential, whereas field-scale balances enable a direct link with criteria for the protection of soil and other environmental concerns. The direct application of process-level information from a geographically small scale (for example, a field) to regional management is often problematic and at times clearly in error. On the regional scale, there are few opportunities to observe and quantify directly the myriad processes and their combinations that influence the water and chemical fluxes on lower scales. The methodology of scale translation therefore demands careful consideration. 'Down-scaling' studies are used to decompose process information (such as remotely sensed data) from the higher level to the lower (that is, top-down), and 'up-scaling', or aggregation, studies use results from a smaller spatial scale to improve the understanding of processes on the regional scale (Wagenet 1996).

Top-down and bottom-up results can be used in a complementary way as part of a 'research chain' in which basic research (using refined models and basic data) and holistic research (using generic models and lumped parameters) are coupled (see Bouma 1997). With the former, investigations specific to sites and farming systems can be carried out, whereas general trends are discovered via the latter. A challenging, but important, task for such a research chain would be to couple environmental effects with economic analyses such that the effects of heavy-metal management in agrosystems could be stated in monetary terms.

Because agrosystems are very complex and are characterized by many interactions, metal transfers in these systems are highly complex as well (Ross 1994). Owing to the dependence on natural conditions and processes, the role of human management in farming processes will always be less efficient than in industrial processes that are not subject to these environmental influences (Pettersson 1993).

Most heavy-metal inputs to agricultural soils originate from atmospheric deposition and from different soil additives (for example, sewage sludge, commercial nitrogen fertilizers, lime, pesticides, manures and compost). A distinction can be made between intentional inputs (such as Cu compounds used as a pesticide or fertilizer) and unintentional inputs (for example Cu as a constituent of soil additives, such as manures). Purchased and farm-processed materials can be controlled directly, to some extent. However, unintentional and uncontrollable inputs into agrosystems include atmospheric deposition and sedimentation after inundation in areas that are regularly flooded. These require different control strategies.

The contaminant burden of a crop is usually a combination of interception, surface contamination from adhering soil and uptake from the soil via the root system. In the field, all processes may operate at the same time. Hence total metal contents in the soil and concentrations in the plant do not always correlate. Root uptake and internal metabolism are important processes, but the contamination of plant surfaces with soil particles may also be important in some cases.

Movements of metals from soil to the (external) environment, other than in harvested crops, are mainly caused by leaching to groundwater and run-off to surface waters. The total amount of heavy metals in soil is the sum of fixed, adsorbed and dissolved heavy metals. The leaching rate from the plow layer is related to the soluble fraction of the total soil metal content.

A 'static balance' is comparable to a black box model that serves to find relationships between the input and output of a system without knowing the system's structure and behavior. In a static balance, a record is kept of the input and output flows, where the output flows are assumed to be unrelated to the metal content (stock) in the soil. The change in heavy-metal content in the plow layer is therefore the result of the net difference between input rate and (constant) output rates. Because a static balance does not consider the dependence between soil content and output flows, it cannot realistically simulate the heavy-metal soil content over time.

As an illustration, the static balances of Cd, Cu, Pb and Zn in different Dutch experimental farming systems on the same location presented in Table 33.2 are shown as the sum of inputs (fertilizers and deposition) minus outputs (leaching and crop offtake).

For simulating the fate of metals in time, a 'dynamic balance' may be calculated in which the relationship between heavy-metal soil content and output flows is explicitly included. A dynamic balance relates the output rates (for example, by leaching and crop uptake) to the metal content of the soil. Further details on static and dynamic balances are provided by Moolenaar *et al.* (1997a). Dynamic heavy-metal balances on the field scale take into account accumulation, leaching and uptake. Such balances can identify 'hot spots' regarding specific crops and metals among and within various subsystems.

With respect to long-term projection, the dynamic balances of Cd and Cu, which represent extreme cases in the conventional arable farming system with mineral fertilizers only (average of system with mineral fertilizer only and system with mineral plus organic

Table 33.2 Static Cd, Cu, Pb and Zn balances for arable farming systems at the Nagele experimental farm (grams per hectare per year)

Arable farming system	Cd	Cu	Pb	Zn
Ecological	1.66	104	80.9	464.5
Integrated	0.75	53.3	37.6	702
Conventional; mineral fertilizer only	0.96	53.5	46.3	804
Conventional; mineral plus organic fertilizer	4.93	−25	32	55

Source: Moolenaar and Lexmond (1998).

fertilizer (see Table 33.2), were calculated for the conventional farming system as a whole (CAFS-total). The values for the input rates, removal rates by harvest, and leaching rates were substituted in the dynamic balance equations. With these rates the development of soil content, leaching and uptake was projected. Soil Cd content would increase from 0.5mg/kg to exceed the Dutch reference value for this soil (0.6mg/kg) and to increase further towards a steady-state value of about 1.7mg/kg. The leaching rate would increase from about 0 to 0.9 grams per hectare per year (g/[ha.yr]) while the offtake rate would increase from about 0.8 to 4.9g/[ha.yr]. These numbers are illustrated in Figure 33.1. The high average Cd offtake rate would result in quality standards of some crops being exceeded, as is shown in the next section, on sustainability indicators.

Figure 33.2 shows that the copper soil content decreases from 60 to less than 14mg/kg at steady state, with associated reductions in leaching and crop offtake rates from 8 to 4

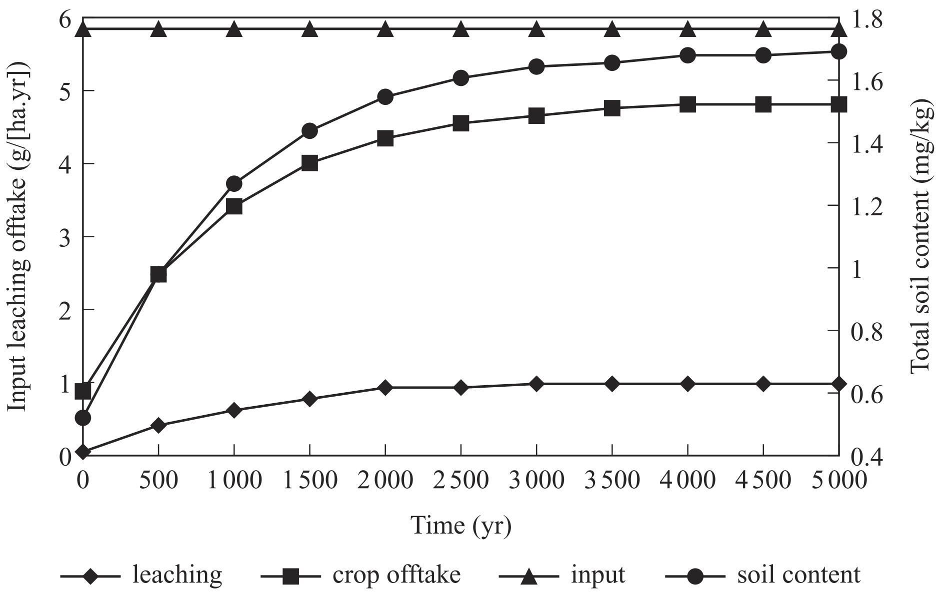


Figure 33.1 Development of cadmium input and soil content, leaching and offtake rates in the conventional arable farming system

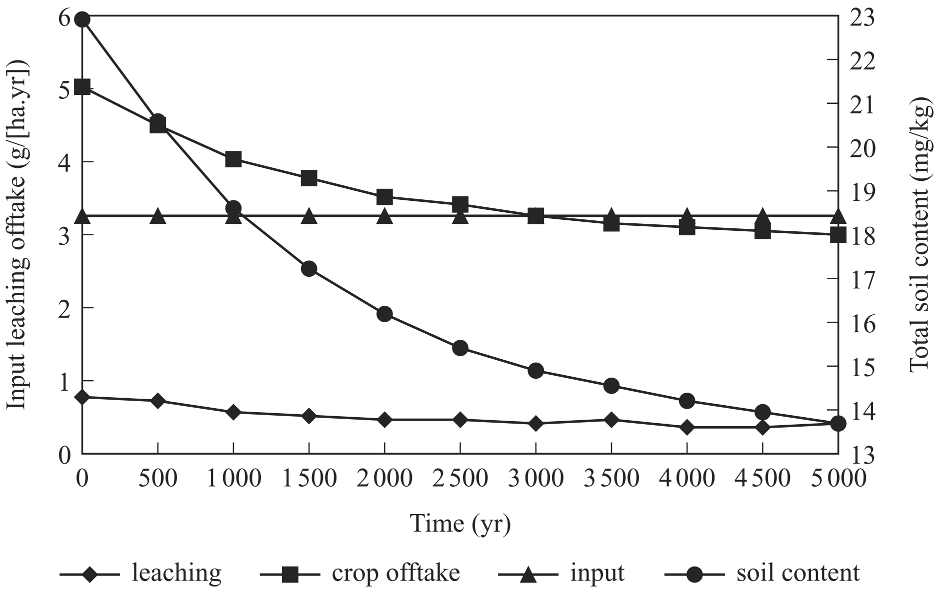


Figure 33.2 Development of copper input and soil content, leaching and offtake rates in the conventional arable farming system

and 50 to 28g/[ha.yr], respectively. So, in the long run, cadmium levels exceed soil quality standards, whereas copper is depleted on this specific soil for this specific farming system.

If an element is naturally abundant (for example, iron and aluminum) and accumulates in a chemical form with solubility similar to its naturally occurring compounds, accumulation may proceed without any appreciable effect on the system itself, or the environment. However, accumulation of Cd, Cu, Pb and Zn mostly involves a steady increase in activity and/or mobility in the soil (van Riemsdijk *et al.* 1987). The rate of this increase depends on the soil's buffering capacity and on the actual input surplus on the balance sheet. Accumulation of these elements thus leads to increased flows between the system compartments, increased contamination of produce and increased leaching and run-off. The dynamics are determined by the metal's solubility in water, their chemical reactivity, and the physical and chemical environment. This results in different cycling characteristics and balances for different elements (Frissel 1978; van Riemsdijk *et al* 1987).

Another issue is also very important for substance flow analysis (SFA) studies. The heavy metals of concern are mostly components of bulk materials that serve as their carriers. To study certain substances it is thus necessary to carry out a material flow analysis (MFA) at the same time. For example, Cd may be carried both by P fertilizers and by refuse compost. Owing to the great differences in the physical matrices of these materials, these different types of carriers influence the form, rate and final steady state of Cd accumulation in soil. For details see Moolenaar *et al.* (1997a). Long-term speciation changes, with impacts on soil organic content and pH, can also be taken into account with the aid of speciation models (Moolenaar *et al.* 1998).

SUSTAINABILITY INDICATORS

As noted, parameters related to plant uptake and leaching depend on many chemical, physical and biological properties of the soil–plant system. The combination of a huge variety of soil properties and these chemical, physical and biological conditions makes the development of general rules for quantitative evaluation of soil quality a difficult task. Fortunately, several characteristic numbers can be derived from input and output rate parameters, in relation to existing or proposed quality standards. They serve as useful indicators to quantify (adverse) effects on the soil and related compartments.

Discrepancy Factor

This indicator compares the input rate for a well-defined system with the total acceptable output rate for that system. It is based on existing or proposed standards for acceptable crop quality and groundwater quality. If the overall input rate exceeds the sum of allowable output rates the discrepancy factor exceeds unity and problems are expected to occur. By comparing the discrepancy factors for different metals we can assess which heavy metal will eventually lead to the largest violation of (groundwater or crop) standards; that is, which metal is relatively most 'abundant'. This indicator allows for prioritization between different metals. The discrepancy factor for the soil compartment may thus be defined as:

$$F_d = \frac{A}{U_c + L_c},$$

where

F_d = discrepancy factor,
A = inputs from various sources,
U_c = maximum acceptable removal rate by harvest,
L_c = maximum acceptable leaching rate.

The value of the discrepancy factor may underestimate the real discrepancy between input and acceptable output since it uses the summation of U_c and L_c. In practice, one of these two removal rates determines which heavy-metal input is still acceptable. Moreover, the discrepancy factor does not take into account any standards for soil ecology. Therefore the value of F_d serves as a first indicator of potential problems only.

Sustainability Factors

If limited data regarding water flow, heavy-metal sorption, mobility and bio-availability are available, a more advanced assessment is feasible. The ranking of the threat to the different compartments of the system, at steady state, depends on which limit will eventually be exceeded by their greatest margin (but not necessarily first). The degree of threat can be assessed relatively, however, by comparing the steady-state values of the soil content, the crop uptake rate and the leaching rate with the corresponding critical values. The corresponding ratios can be regarded as sustainability factors for ecology (F_e), crop uptake (F_u) and soil solution or leaching (F_s).

The *critical* sustainability factor (F_c) is given by:

$$F_c = MAX(F_e, F_u, F_s).$$

Sustainability Times

Although the indicators described thus far are all derived from the dynamic balance they do not yield information on the time horizon when standards will be violated. Comparing sustainability times identifies the compartment for which the quality standard is exceeded soonest, assuming continuation of current conditions. The *critical* sustainability time (t_c) identifies the compartment for which the quality standard (soil, crop or groundwater), if exceeded, is exceeded first and is thus defined as:

$$t_c = MIN(t_e, t_u, t_s),$$

where

t_e = time at which the ecological quality standard is exceeded,
t_u = time at which the crop quality limit is exceeded,
t_s = time at which the groundwater limit is exceeded.

All sustainability indicators are based on dynamic balances and they enable screening and comparing of different agro-ecosystems without having to know all processes in detail.

A study of the same arable farming systems that were used to illustrate the static and dynamic balances is used here to illustrate the sustainability indicators (Moolenaar *et al.* 1997b). In Table 33.3 the values for the (critical) soil cadmium contents, the (critical) sustainability factors and the (critical) sustainability times are presented for the four systems.

Table 33.3 Sustainability indicators of four arable farming systems

	Ecological	Integrated	Conventional	
			organic	mineral
G_{ss}*	1.31	0.80	0.94	1.74
F_d	1.28	0.94	0.94	2.44
F_e	4.00	1.67	2.13	4.73
F_u	2.29	1.60	1.53	3.57
F_s	0.01	0.13	0.21	1.03
t_e	145	306	245	70
t_u	622	362	696	153
t_s				3 366

Note: * G_{ss} is the steady-state soil cadmium content in mg/kg.

Comparing the discrepancy factors (F_d) of these systems already gives an idea of the most sustainable options. For the ecological and conventional mineral fertilizer (MF) systems, the F_d value is larger than 1 and for the integrated and conventional organic and mineral (OF) systems it is smaller than 1. Based on the discrepancy factor, most problems are expected for the conventional MF system and no real problems are expected for the integrated and conventional OF systems. The values of the sustainability factors are good indicators of the most threatened compartment at steady state. For all systems $F_e > F_u > F_s$ and thus F_c equals F_e. This suggests that, at steady state, ecology is threatened most and groundwater least for current standards. According to the sustainability times, all standards will be exceeded for the conventional MF system, but the time span varies between 70 and more than 3000 years for ecology and groundwater, respectively. Although the F_d value is smaller than 1 for the integrated and conventional OF systems, in both systems the F_u value is larger than 1 and the crop quality standards will be exceeded. These results illustrate that for discrepancy factors close to unity the effect for the least threatened object (leaching in this case) may mask the violation of the standard for the other output term (here uptake). Thus neither for the integrated nor for the conventional OF system does the discrepancy factor identify the violation of crop quality standards. Moreover, the discrepancy factor does not identify any problems for ecology, while the F_e values are largest in every system. Hence, even for F_d values of order 1 it may be worthwhile to estimate the sustainability factors in addition to the discrepancy factor if the required information is available.

Another field of application may be the incorporation of sustainability indices in a generic data set such as a Geographical Information System (GIS). If the parameters in a GIS could be combined in such a way that the rate parameters were known, this could be very promising for larger-scale (for example, whole region) assessments. As an intermediate approach, Guinée *et al.* (1999) used national statistical (averaged) data on different farming systems and fertilizer applications to quantify Cd flows and accumulations in the agricultural sector as a whole. An aggregated analysis at the national level was carried out by defining representative crop–soil combinations. For each such combination, the rate parameters can be estimated and, in this way, large-scale scenarios can be analyzed.

Both non-essential elements (like Cd) and essential elements (like Cu and Zn) with a nutritional function and potential deficiency problems for crop production may be taken into account. Both depletion and accumulation are related to sustainable agricultural practices and hence to the sustainability indicators.

PERSPECTIVES FOR SUSTAINABLE HEAVY-METAL MANAGEMENT IN AGROSYSTEMS

Different strategies for heavy-metal management will have different consequences for the resulting steady state. Short-term strategies may aim at increasing the soil's buffer capacity. For soils with a low sorption (or buffering) capacity, metal concentrations tend to increase in groundwater and crops. This may result in groundwater and crop quality standards being exceeded, but without significant accumulation in the soil. Good management practice might attempt to lower the crop uptake rate, for example by raising the organic matter content (increasing retention), favoring competition by applying calcium and magnesium and competing heavy metals. Changing tillage practices may influence the stratification of pH, organic matter and metals. Moreover, cultivars may be changed and acid soils may be limed to increase the pH (see, for example, McLaughlin *et al.,* 1994, 1995).

Some practices that aim at lowering crop uptake (such as stimulating competition) may at the same time lead to higher leaching rates or vice versa, thus resulting in a trade-off between leaching and uptake. Moreover, minimizing output rates by management practices will result in the steady-state content being reached later at a higher level. Selecting cultivated crops with pronounced heavy-metal removal (within critical limits) can be very sensible for farming systems with low input. A higher crop uptake rate results in a lower steady-state soil content due to less accumulation.

Long-term strategies focus on reducing inputs to soils. This results in the steady state being reached with lower total accumulation and lower output rates. Input reduction can be achieved by reducing the amount of heavy metals in source material (quality) and by reducing the amount of fertilizer or manure added to the soil (quantity). This kind of input reduction could be aimed at by decreasing application (for example, by educating farmers on how to use nutrient and heavy-metal balances) or by changing the production system (for example, to a mixed farming system).

Industrial ecology suggests that economic actors (including farms as well as firms) should optimize the use of energy and material, minimize waste production and use the effluents of one process as input for another process (Frosch and Gallopoulos 1989). Encouraging symbiosis, or relationships of mutual benefit, is evidently one key to industrial ecology. There is evidence that mixed farming systems compare favorably with specialized (for example, grain or dairy) farming systems with regard to heavy-metal accumulation. Owing to the internal cycling of forage and manure, fewer external inputs are required and thus imports of heavy metal-containing raw materials and products are minimized (Moolenaar and Lexmond 1998).

Mixed farming need not be restricted to the farm level. Optimization of energy and material use and minimization of waste production may be enhanced by exchanging intermediate outputs between two or more specialized farms. To test the hypothesis that such a mixed farming system might improve sustainability of agriculture, Bos and van de Ven (1999) carried out an interesting study in the Dutch province of Flevoland. They quantified nutrient balances, labor requirements and labor income for a specialized grain farm, a specialized dairy farm and both combined into a mixed farming system, all exchanging land, labor and machinery. It was concluded that in a mixed farming system it is possible to realize a higher income and to reach higher production levels without increasing environmental pollution. Prospects for mixed farming systems at the regional level depend on the future balance between integration (to enhance sustainability) and specialization.

CONCLUSIONS AND RECOMMENDATIONS

Heavy-metal balances can be used to determine the options for a sustainable heavy-metal management in agriculture. These balances are a means to quantify heavy-metal (input and output) flows and the resulting accumulation. Heavy-metal flows in agrosystems can be studied within the broader context of substance flow analysis and industrial ecology on a variety of scales and for different systems.

Heavy-metal balances on a national scale provide valuable information for economic analyses. Generic balance studies indicate that legislation limiting metal emissions by industry and using fertilizers with low metal content may be very important. Whereas policies based on direct economic instruments or generic regulations often ignore farm characteristics and individual management options, field-scale and farm-gate balances give farmers specific feedback on effective options for better heavy-metal management. Farm-gate balances show the total contaminating potential, whereas field-scale balances enable a direct link with criteria for the protection of soil and other environmental criteria.

Dynamic heavy-metal balances on the field scale taking into account accumulation, leaching and uptake can identify 'hot spots' regarding specific crops and applications for different metals among and within various systems. The dynamic balance approach thus proves to be a useful tool with which to compare the heavy-metal management of agrosystems.

Although data from long-term field experiments are needed to study the long-term environmental consequences of applying fertilizers and soil conditioners, they may not give sufficient information because variation in data collection may hamper the reliability of data and it may be difficult to maintain relevance for current agricultural practices owing to changing practices, technologies, cultivars and natural variation. Therefore, in addition to monitoring programs, projection models are needed to assess environmental consequences of different management practices. Dynamic heavy-metal balances are useful for projection purposes.

Perspectives for sustainable heavy-metal management in agrosystems are as follows:

- monitoring strategies using representative agrosystems (experimental farms) and adequate monitoring networks;
- quantification of uncertainties;
- dynamization and incorporation of speciation and environmental fate and effect modeling in balance studies;
- scale aspects (compatibility of SFA on different scales: 'up-scaling' and 'down-scaling');
- sustainability indicators (socioeconomic, ethical, environmental, agronomic aspects and integrated approaches);
- mixed farming systems (using the potential of a region);
- coherent international policies (covenants, trade barriers and so on).

34. Industrial ecology and automotive systems

Thomas E. Graedel, Yusuke Kakizawa and Michael Jensen

THE AUTOMOTIVE TECHNOLOGY SYSTEM

In many ways, the motor car illustrates the depth and breadth of industrial ecology. The materials from which it is made can be studied (for example, Gibson 2000). Its individual components – instrument panels, front ends and so on – can be analyzed in detail (for example, Ryding *et al*. 1993; Keoleian 1998). And the entire car has been the subject of a variety of industrial ecology analyses (for example, MacLean and Lave 1998; Sullivan *et al*. 1998). All of these approaches provide information useful for improving environmental performance. Nonetheless, they tend to avoid facing a central, vitally important fact: cars and their use are embedded in and are products of cultural systems.

A system is a collection of interacting, interdependent parts linked together by exchanges of energy, matter, and/or information. In the case of the car and its related technological and societal structures, the system is that pictured in Figure 34.1 (Graedel and Allenby 1998). At its lowest level, it comprises the technology-rich mechanical subsystems and the manufacturing processes by which they are made. These subsystems and processes have been a predominant focus of environmental regulation, but, taken as a whole, they are probably not the major contributors to the car's environmental impact, except in a very local sense. For example, it is certainly reasonable to encourage the use of paints that do not contribute to local air pollution (for example, paints that have low emissions of volatile organic compounds), but at the same time one should recognize that paint emissions are not a major environmental impact of the motor car. The same is true of the next level of complexity, the entire car itself.

The third level, car use, is more important. There are two major dimensions to this system level: technical and cultural. On the technical side, great progress in reducing environmental impacts has been achieved, and more is possible. Examples include sensor systems that monitor oil composition and properties and recommend oil changes only when needed, and sensor systems that report when air pressure in tires is low (low pressure results in higher gasoline consumption and greater tire wear). From the cultural standpoint, however, failure to address environmental impacts is virtually total (Derr 1995; Bauer 1996). The mix of cars purchased in more developed countries (and soon to be available in rapidly developing countries such as China) and the ways in which they are used are increasingly inefficient. In North America, families routinely purchase vehicles with four-wheel-drive systems and high gasoline consumption and drive them farther and farther each year (USFHA 1998). In Germany, attempts to impose any speed limits on the autobahn routinely fail, even though the German environmental party (the Greens)

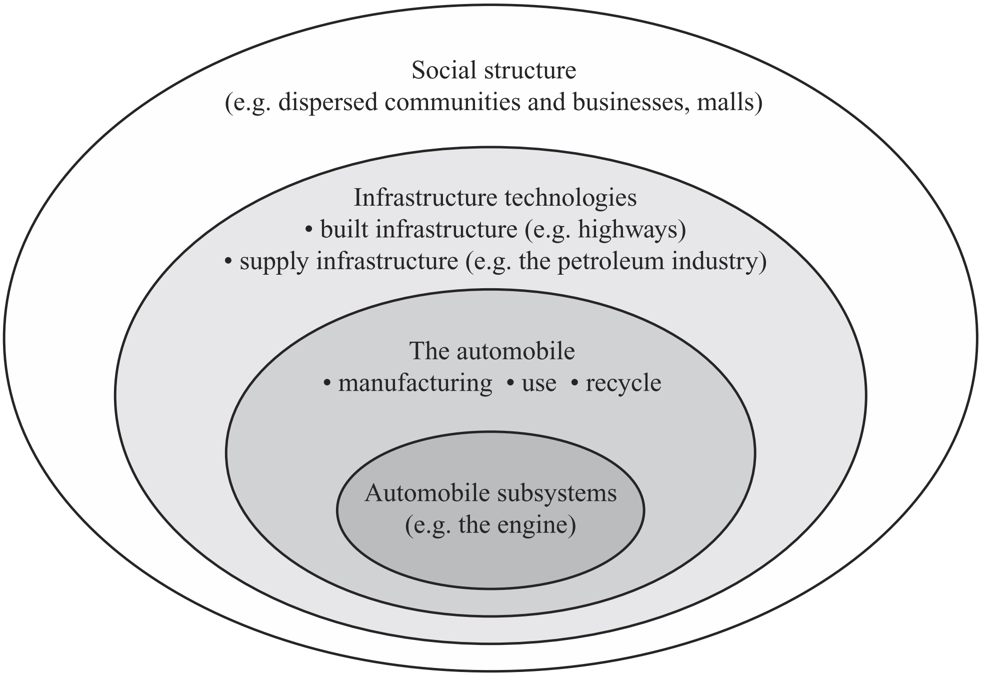


Figure 34.1 The automotive technology system: a schematic diagram

is widely popular. Even a cursory evaluation of this simple systems model, therefore, indicates that much attention is being focused on the wrong subsystem, and illustrates the fundamental truth that a strictly technological solution is unlikely to fully mitigate a challenge that is partly cultural.

The most significant environmental impacts of the automobile technology system thus arise from its higher, not its lower, levels. Consider the energy and environmental impacts that result from just two of the major infrastructures required by the use of cars. The construction and maintenance of the 'built' infrastructure – the roads and highways, the bridges and tunnels, the garages and parking lots – involve huge environmental impacts. The energy required to build and maintain such infrastructure, the natural areas that must be perturbed or destroyed in the process, the amount of materials required, from aggregate to fill to asphalt – all of this is required by the automobile culture, and attributable to it. Similarly, the primary customer for the entire petroleum sector – and, therefore, causative agent for much of its environmental impacts – is the car.

The final and most fundamental effect of the car, however, may be in the geographical patterns of population distribution to which it has been a primary contributor. Particularly in lightly populated and highly developed countries such as Canada and Australia, the car has resulted in a diffuse pattern of residential and business development that is unsustainable without constant reliance on the car. Lack of sufficient population density along potential mass transit corridors makes public transport uneconomic within many such areas, even where absolute population density would seem to augur otherwise. This transport infrastructure pattern, once established, is highly resistant to change in the short term, if for no other reason than that residences and commercial buildings last for decades. Thus high demand for personal transport (that is, the car) is firmly embedded in the physical structure of the community.

AUTOMOTIVE SYSTEM LIFE CYCLES

For each level of the automotive system, the entire life cycle should be considered. That is, one should consider the extraction from their reservoirs of the materials that are used, what happens to them (and the environment) during product manufacture, how the use of the product or system affects the region within which the use occurs and, finally, what happens to the product or system and its materials once it is obsolete or the consumer disposes of it.

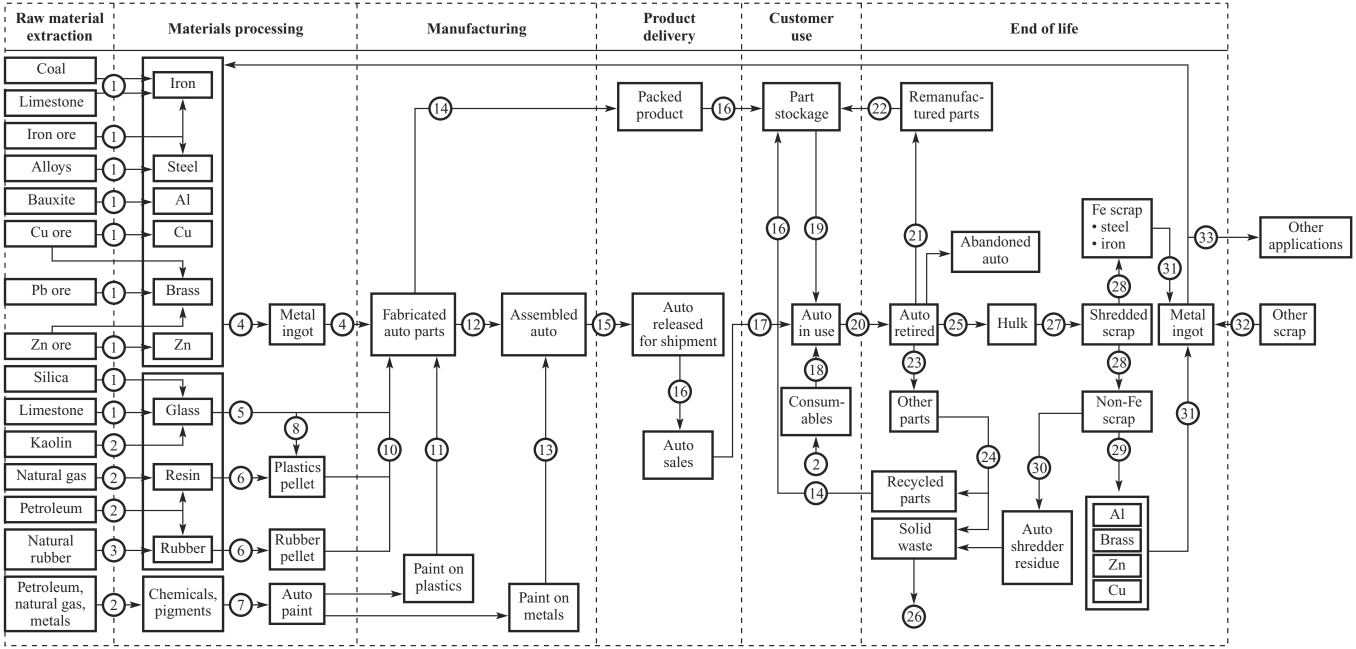
The life cycle of a car itself can be put into perspective with the help of Figure 34.2 (Graedel and Allenby 1998). The cycle begins with the extraction of raw materials either from virgin sources or from a recycling stream: petroleum, coal, metal ores, rubber and so on. The raw materials are then processed by a variety of industrial techniques to produce finished materials suitable for use in manufacture. During manufacture, another series of processes bend and shape the metal, mold the plastic, and perform the other operations needed to create components and assemblies from the finished materials. Product completion is followed by packaging, delivery and customer use, generally for extended periods of time. A variety of maintenance activities require additional material streams: lubricants, coolants, tires and so on. When the vehicle is finally retired, an extensive recycling operation returns a high percentage of the materials back into the industrial materials supply system. The entire cycle for a typical vehicle takes 10–14 years.

Similarly, one can construct a diagram (Figure 34.3) of the life cycle of the automotive infrastructure system. The form of Figures 34.2 and 34.3, adapted from Keoleian *et al.* (1998), is obviously similar, though there are differences in processes and products. Another difference is the time scale involved for the infrastructure life cycle. It is obviously much longer than for a car; bridges have lifetimes of perhaps half a century or more, and many road surfaces are periodically repaired and used essentially forever.

Thus all products and systems can be perceived and evaluated from the viewpoint of their life cycle. There are several crucial insights that result from such an approach. One is that the designer should view the product or system, not from the standpoint of 'out of sight, out of mind', but over a period that, including the customer use and end-of-life stages, may encompass decades. Another is that one should consider the life cycle impacts of the choices of materials, since their extraction and processing stages can have very large environmental effects. A third is that the end-of-life stage can be very much enabled or disabled by the decisions made by the design team concerning which materials are selected or which assembly techniques are used.

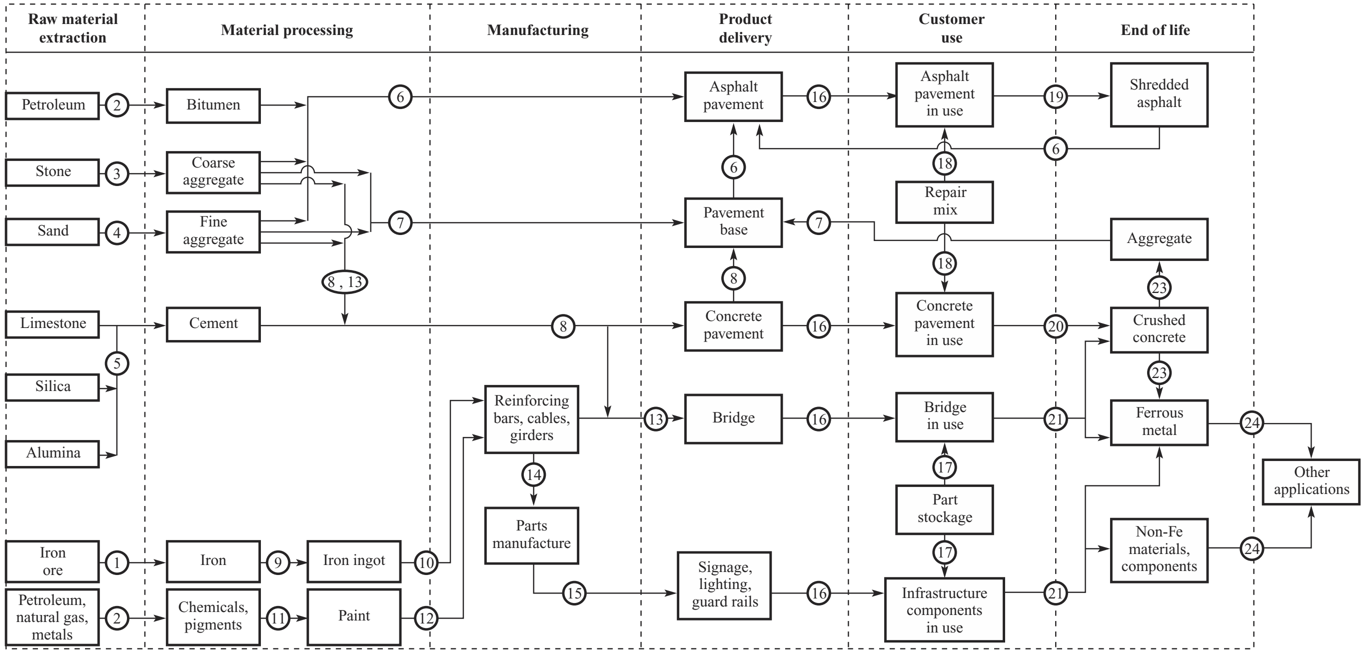
ASSESSING AUTOMOTIVE ENVIRONMENTAL PERFORMANCE, 1920–99

Comprehensive life cycle assessment (LCA) is discussed in Chapter 12 by Udo de Haes. Although comprehensive LCAs have not been conducted for motor cars, the life cycle



1. Mining, refining, ore processing
2. Drilling, distillation, processing
3. Latex gathering, processing
4. Melting, casting
5. Melting, refining, processing
6. Molding
7. Paint manufacturing
8. Compounding
9. Casting, extrusion, stamping, machining, joining
10. Blow molding, injection molding
11. Spraying, in-mold coloring
12. Auto assembly
13. Spraying
14. Packaging
15. Preparing auto for shipment
16. Delivery to dealers
17. Delivery to customer
18. Use of consumables
19. Parts sale
20. Sale of junk auto
21. Re-use/ remanufacture
22. Sale of remanufactured parts
23. Dismantling
24. Recycling
25. Sale, auto or other applications
26. Landfill disposal
27. Shredding
28. Magnetic separation
29. Separation of non-Fe metals
30. Air separation
31. Processing of scrap materials
32. Mixing of other scrap
33. Manufacturing of other products

Figure 34.2 The life cycle of the motor car, and the processes that occur during that cycle



1 Mining, refining, ore processing
2 Drilling, distillation, processing
3 Crushing
4 Sieving
5 Cement manufacturing
6 Asphalt pavement
7 Pavement base manufacturing
8 Concrete pavement manufacturing
9 Melting, casting
10 Casting, forming, machining
11 Paint manufacturing
12 Spraying
13 Bridge manufacturing
14 Stamping, shaping, joining
15 Infrastructure component manufacturing
16 Release for service
17 Bridge and component manufacturing
18 Pavement patching
19 Asphalt shredding
20 Concrete crushing
21 Bridge disassembly and demolition
22 Infrastructure disassembly and recycling
23 Magnetic separation
24 Manufacture of other products

Figure 34.3 The life cycle of the automotive infrastructure, and the processes that occur during that cycle inventory (LCI) component of LCA has been applied to cars and car families with some success (Sullivan *et al.* 1998; Finkbeiner *et al.* 2000). However, a variety of uncertainties and assumptions render the results somewhat problematic (Keoleian *et al.* 1998; Lave *et al.* 1998; Saur *et al.* 1998). To improve the efficiency and communicability of LCAs, streamlined LCAs (SLCAs) have been developed over the past several years (for example, Graedel 1998) and can readily be applied to cars. In the approach we will use here, the SLCA assessment system has as its central feature a 5×5 assessment matrix, one dimension of which is life cycle stages and the other of which is environmental stressors. (In the present case, the life stages are defined as pre-manufacture, manufacture, product delivery, product use and end of life. The environmental stressors chosen are materials choice, energy use, solid residue generation, liquid residue generation and gaseous residue generation.) To use the SLCA matrix tool, the assessor studies the product design, manufacture, packaging, in-use environment and likely disposal scenario and assigns to each element of the matrix an integer rating from 0 (highest impact, a very negative evaluation) to 4 (lowest impact, an exemplary evaluation). In essence, what the assessor is doing is providing a figure of merit to approximate the result of the more formal LCA inventory analysis and impact analysis stages. She or he is guided in this task by experience, a design and manufacturing survey, appropriate checklists and other information.

As an additional aid to analysis and interpretation, 'target plots' can be constructed. In these displays, the value of each element of the matrix is plotted at a specific angle. (For a 25-element matrix, the angle spacing is $360/25 = 14.4°$.) A good product or process shows up as a series of dots bunched toward the center, as would occur on a rifle target in which each shot was aimed accurately. The plot makes it easy to single out points far removed from the center and to mark their topics out for special attention by the design team. Furthermore, target plots for alternative or evolutionary designs of the same product permit quick comparisons of environmental attributes.

Model T Fords were produced in the 1920s at the River Rouge Plant south of Detroit. Henry Ford's policy was to centralize manufacture and to control the supply of materials needed. As a result, the Ford Company owned iron mines, forests, a railroad, a glass plant and Great Lakes shipping companies. The Rouge Plant to which this material flowed was the largest manufacturing facility in the world. It provided substantial opportunities for environmental degradation, but Henry Ford's obsession with avoiding waste (Jensen 2000) resulted in a number of recycling activities that we would today call forward-looking: recovery of coke oven gases and use of wood scraps for fuel, for example. The vehicles themselves were evolutions of horse-drawn carriage design, and utilized relatively simple materials and processes.

By the 1950s, manufacturing activities were decentralized, massive vehicles with little recycling potential were common, and Henry Ford's legacy of waste minimization had largely been forgotten. Emissions to air, water and soil were substantial, and recycling activities were generally absent.

The 1980s saw major changes. Design attention was focused on the manufacturing and product use life stages, and environmental regulations and the oil crisis encouraged much-improved emissions performance and on-road fuel efficiency.

By 1998, the ubiquitous presence of automotive electronics had further improved the efficiency and reliability of vehicles, and pollution prevention programs in manufacturing were reducing environmental impacts at that life stage as well. Many plastic parts were being made of salvaged material. Much effort was being made to design vehicles with recycling in mind.

During this 80-year period, the materials from which cars were made evolved substantially. As with the carriages and wagons that provided much of the inspiration for early designs, the 1920s car frame was largely of iron, the engine and control linkages of steel, and the body of wood. By weight, early vehicles were about three-fourths iron and steel, with wood, seats, tires and fluids making up most of the remainder. By the 1950s, wood was no longer in use, and the amount of iron and steel contained in the car had doubled. Materials used in the mid-1980s were very different: iron and steel content per vehicle was almost back to the 1920s level as cars became smaller. Some of that loss was compensated for by aluminum and plastic, however, both of which were in the 90kg per vehicle range. The late 1990s vehicles were heavier than those of a decade earlier; much of this increase comes from enhanced control, safety and convenience features, though increasing vehicle size was a factor as well.

Making use of the information described above, streamlined life cycle assessments have been conducted on the Ford sedans from four eras. In this section the summary information derived from the comprehensive assessments will be presented and commented upon. Figure 34.4 shows the results for each of the four vehicles, presented individually for each of the five life stages and then collectively over the entire lives of the vehicles.

Recall that the maximum score for each matrix element is four and, since each life stage has five different environmental concerns, the maximum score for each life stage is 20. The top panel in Figure 34.4 evaluates life stage 1, pre-manufacture. Here the 1920s and 1998 vehicles are essentially equivalent at about 10 out of 20, while the 1950s vehicle scores about half as well and the 1980s vehicle only slightly better. In general, the higher scores for the 1920s vehicle reflect the use primarily of benign materials and of highly integrated manufacturing. Those for the 1998 vehicle reflect intensive use of recycled and energy-efficient materials, and of close supervision of the environmental attributes of supplier components.

The manufacturing life stage is shown in the second panel of Figure 34.4. The 1998 vehicle clearly scores better than the others, and is more than three times better than the 1950s vehicle. As with life stage 1, the 1920s vehicle ranks second, the 1980s vehicle third. Manufacturing in the 1920s involved little in the way of chemicals, and re-use of scrap materials was extensively practiced. A steady increase in scoring (from a low start) occurred after the 1950s. Among the reasons are the introduction of CFC solvents in the 1940s (and their eventual elimination by the 1990s), the increasing control of volatile organic carbon compounds (VOCs) from painting operations, and the increasing amounts of recycling that occurred toward the end of the 20th century.

The third panel – the packaging and shipping life stage – continues to show the 1998 vehicle as environmentally superior. The rest are bunched more closely than before, with the 1920s and 1980s vehicles essentially equal. The scoring essentially reflects the short time and small environmental impacts that occur during the product delivery stage, and the fact that approaches to delivering vehicles have changed less over time than have approaches to other life stages.

The product use stage reflects principally the degree of exhaust emissions control implemented on the vehicles. As a result, the 1998 vehicle has the highest score, the 1980s vehicle the next highest. The 1920s vehicle scores higher than the 1950s vehicle, not because of

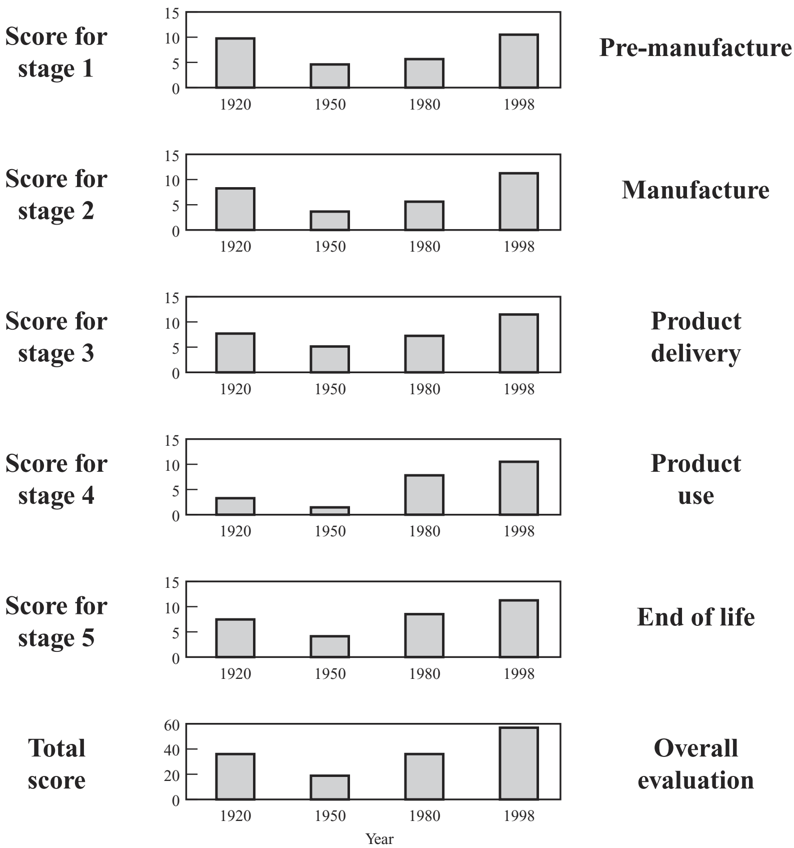


Figure 34.4 The results of the SLCA assessments for each of four cars from different epochs over the five life stages, and the overall assessments

exhaust gas treatment, but because its smaller size and smaller engine generated lower volumes of exhaust pollutants.

The fifth life stage assessment includes impacts during product refurbishment and as a consequence of the eventual discarding of modules or components deemed impossible or too costly to recycle. Most modern cars are recycled (some 95 per cent currently enter the recycling system) and from these approximately 75 per cent by weight is recoverable for used parts or returned to secondary metals dealers. There is a viable used parts market and most cars are stripped of re-usable parts before they are discarded. Improvements in recovery technology have made it easier and more profitable to separate the car into its component materials.

In contrast to the 1950s, at least two aspects of modern car design and construction are retrogressive from the standpoint of their environmental implications. One is the increased diversity of materials used, mainly the increased use of a variety of plastics but also an increased diversity of alloys. The second aspect is the increased use of welding in the manufacturing process. In the vehicles of the 1950s, a body-on-frame construction was used. This approach was later switched to a unibody construction technique in which the body panels are integrated with the chassis. Unibody construction requires about four times as much welding as does body-on-frame construction, plus substantially increased use of adhesives. The result is a vehicle that is stronger, safer and uses less structural material, but which is much less easy to disassemble.

The bottom panel on Figure 34.4 gives the overall assessment for the four vehicles. It clearly shows the 1998 vehicle to be the most environmentally responsible. The 1920s and 1980s vehicles finish in a virtual tie, the 1920s advantage in the earlier life stage being offset by the later life stage superiority of the 1980s vehicle. The 1950s vehicle – large, heavy, manufactured without environmental sensitivity and with no emissions controls – finishes a distant fourth.

The target plot representations of the results are given in Figure 34.5. For the 1920s Model T, the plot indicates relatively good pre-manufacturing, manufacturing and product delivery performance, poor in-use performance and unpromising end-of-life

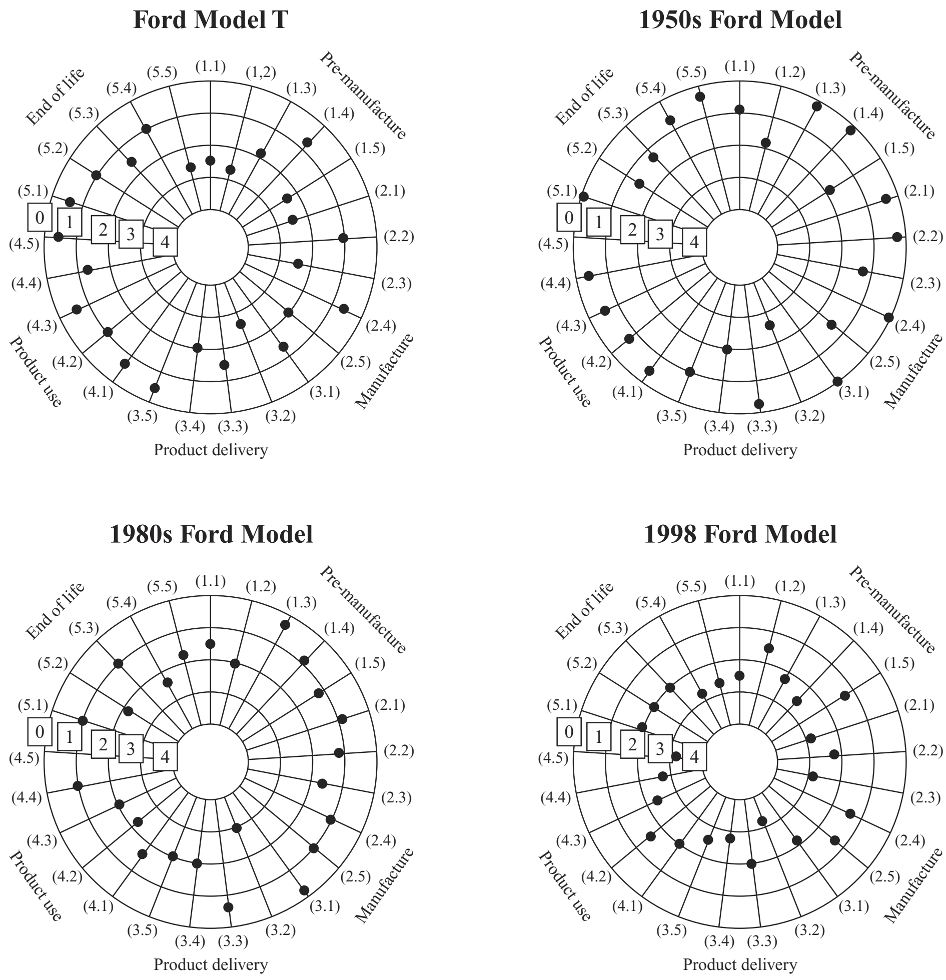


Figure 34.5 Target plots for the environmental assessments of the four cars

ratings. The 1950s vehicle has a target plot with few high-value entries anywhere. In the 1980s, low scores for pre-manufacture and manufacture were offset by better product delivery and in-use scores. The 1998 vehicle, with some deficiencies, shows a fairly high degree of environmental responsibility across environmental stressors and life stages.

Such a conclusion (a decrease in environmental performance during the first half of the 20th century followed by an increase during the second half) would follow as well for the individual components of the vehicles and for the processes by which they were manufactured (Jensen 2000). For the overall automotive system, however, the dispersion produced by modern approaches to urban land use paints a different picture. We explore this topic, and possible alternative strategies, in the following section.

URBAN TRANSPORT SYSTEMS

In thinking about the automotive system, it is useful to itemize the reasons why individuals enjoy owning or having access to cars, and also the reasons why some people have reservations about the 'automotive culture'. What are the qualities we value in our current transport system, and what is it that needs to be changed? To answer these evaluative questions, it is necessary mentally to strip car use of its wide-ranging cultural associations and behavioral traditions, and look at the comparative benefits and liabilities of the present system.

Advantages that the public generally associates with owning and operating a personal car include flexibility and freedom, personal security and comfort, speed and the ability to carry moderate cargo loads conveniently. These characteristics have largely established the car in the dominant role it holds today. However, only the cargo-carrying ability of the private car remains a clear advantage, since in many places the other attractive characteristics of the private car are being compromised. As congestion increases, the perceived benefits of car travel become increasingly more vestigial than real.

Conversely, what are the largest perceived drawbacks to car use? A typical list might include congestion, traffic accidents and risk, air pollution and aesthetic degradation. In addition to compromises on speed, flexibility, safety and health, the road and parking infrastructure results in a landscape dominated by asphalt and concrete: highways, intersections and parking lots. Increasingly, there is a call for a different type of city aesthetic, and a lifestyle spent less within the confines of the individual, shiny metal box termed the private car.

A successful urban transport system will incorporate what people like about cars while minimizing or eliminating what they dislike (MacLean and Lave 2000). There are three general approaches to doing so, requiring successively greater changes in lifestyle and infrastructure. These are first, to improve private transport; second, to provide exemplary public transport; and third, to reduce the need for vehicular transport.

The first alternative, to improve private transport, is the most immediately promising, but is ultimately of limited application in cities. Advances in car and road design have resulted in significant cumulative change, including remarkable improvement in safety devices and emissions since the 1960s. New technologies, such as advanced composite frames and hybrid fuel-cell propulsion, promise even further gains in safety, cleanliness and economy. However, congestion and urban dynamics are more inherent problems for private transport. Smaller commuter cars and intelligent transport systems (that is, electronically monitored and controlled highway use) might alleviate these problems, but they cannot fully resolve them. While dynamic urban areas can grow in density by increasing building height, roads must grow in width to accommodate added residents and their cars – and surface area is in diminishing supply. To solve these problems more completely, the public will be forced to move away from exclusive personal vehicle use in densely inhabited areas.

The second alternative is to establish a public transport system that meets or exceeds the expectations of current car users. Although the latter half of the 20th century has often shown this to be a challenging task, none of the aforementioned qualities of car use are inherently alien to public transit. As dense urban areas continue to expand, so will the need for public transit. Currently, however, public transit is not economical or efficient for less densely built areas, and does not appear likely to replace private transport completely in the immediate future.

The third transport alternative is to minimize the need for vehicular travel in the first place. Communications and information technologies have potential to replace physical travel with digital or electronic transfers, in ways such as telecommuting. Purchases over the Internet require little travel other than that of the regular postal system (or a competing carrier). Conscientious community planners can add travel economy to convenience by locating frequently used services within walking or biking distance of residents. But even careful planning cannot obviate the need for vehicular transport in the active and interconnected functions of a modern city.

Separately, these approaches are incomplete. A satisfactory and viable transport solution must combine and coordinate all three approaches. The following proposal concentrates particularly on the first two aspects, combining personal and public transport to create an integrated, yet multimodal, approach to local passenger transport.

In our approach to metropolitan transport systems, it is helpful to realize that, often, different landscapes are best traversed by different means. In freight shipping, for example, efficient transport might require a combination of ocean freighter, barge, rail or semi-trailer truck to cross various physical landscapes. The daily commuter also crosses landscapes: built environments of varying density and dispersion. These range from the widely dispersed landscape of the suburb, to the dense congestion of major transport arteries, to the level of traffic in an individual city neighborhood.

To optimize human transport, we must delineate the boundaries between these landscapes and choose transit modes suitable for each. This system does not view an urban journey as a single ride, with cars and public transit in competition the entire way. Rather, we take advantage of the fact that the different modes of transit excel in different parts of the commute: cars are unsurpassed as suburban transport, and transit is better suited to higher-density environments. By maximizing the efficiency of each stage of the journey, the system is cumulatively optimized as well.

The general model presented herein (Graedel and Jensen 1999) consists of four modes, each tailored to the dispersion of the area it serves. Suburbs are left for car travel; high-volume transit is left to high-speed public rail; city neighborhoods are serviced by tram. Each mode is connected to the others, but within its zone each is exclusive: trams do not cover suburban neighborhoods, and private cars are prohibited in the city neighborhoods. Although such a system requires riders to transfer between modes, this inconvenience is more than offset by the efficiency of each mode in time, driving effort and urban space requirements.

The backbone of this system is a radial network of high-speed transit, carrying through-traffic with infrequent stops. Two collective and distributive networks are attached to this backbone, one for suburbia and one for the city. In the city, trams with frequent stops connect each neighborhood to the nearest transit terminal. In the suburbs, riders use cars to cover the most dispersed portion of their travel route. Strategically located 'park and ride' stations allow easy and uncongested car access to a rapid transit connector, which transfers riders to a high-speed rail terminal on the periphery of the transit backbone (Figure 34.6).

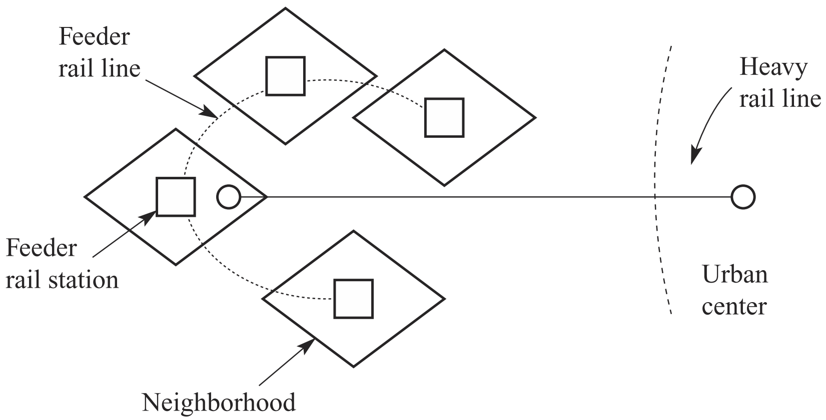


Figure 34.6 A portion of a conceptual transit network for a transmodal system: a web of tram routes serves the urban core

The advantages of such a system mirror the perceived benefits of the automobile. The first is speed. The radial backbone transports passengers more rapidly than city driving speeds, without traffic or the inconvenience of finding a parking place. Second is security and comfort. By eliminating competition between modes, the system could afford comfortable and attractive vehicles, as well as adequate security systems and personnel. The third advantage is flexibility and freedom. Inside the city, the transit system has more complete coverage and requires less passenger forethought than previous systems. Since city transit is split between high-speed and street-side transit, tram routes are shorter and need not overlap. This creates simpler routes, rapid turn-around times, and thus more frequent trips and less waiting.

One function of the car thus far ignored by public transit systems is its ability to move moderate amounts of goods as well as people. Currently, passengers on public transit must carry all items with them, a task that becomes a significant inconvenience even when dealing with such common cargo as groceries. To eliminate personal cars in the inner city, public transit must efficiently fulfill this cargo-carrying function, now so well satisfied by private vehicles.

A common, if imperfect, example of such service is the airline industry, which transfers coded luggage between aircraft until arrival at the final destination. In contrast to airlines, however, public transit must quickly handle larger numbers of smaller transfers. This requires standardized bins, electronic coding and mechanized handling between transit vehicles.

Despite its complexity and cost, a cargo option opens many possibilities to simplify the movement of people and goods in the city. One can envision a city where retail stores transfer goods directly from checkout to a coded bin on the same public transit vehicle that will transport the customer. Purchases could be automatically transferred all the way to the neighborhood stop, or, for suburbanites, to a park-and-ride station, where they would be loaded into the car for the drive home. Public transit might also be designed to accommodate postal and courier services, leaving the remaining urban road space to be used exclusively by commercial delivery trucks, police, fire and ambulance services – thereby shortening vital response times.

Our conclusion is that industrial ecology has, as a field, overemphasized cars as products and underemphasized the transport system of which the car is such a major part. An emphasis on private vehicles is easier and more familiar for technologists, but will almost certainly result in unsustainable systems over the long term. In our vision, industrial ecology should address the overarching concepts and assumptions of local transport systems as well as carefully designing its details. Rather than modifying an increasingly obsolete system, we must salvage its strong points and create a new and improved response to the need for personal transit.

A wide and far-sighted view is necessary both for planning and for implementation. Cost and commitment, not technology, are the greatest obstacles to a restrictive multimodal system. Technology that satisfies the requirements of the four outlined modes already exists, but broad public support does not. The difficult commitment to abrogate the motor car within the entire central city must be supported both initially and throughout the network's development. The most critical link is the performance and acceptance of the public transit elements. By providing speed, flexibility, security, cargo space and comprehensive coverage of the initial 'no-car' zone the transit system would retain and build the support that will be required (Ausubel *et al.* 1998).

The process of creating a transport paradigm for the future is demanding. The pace and nature of implementation must be designed with social considerations in mind, since changing our metropolitan transport networks requires accompanying changes in lifestyle. The transition must be carefully planned and made comprehensible and compelling to all. In addition, system designers must provide for the future by building in the capability for adjusting to the changing demands of a dynamic city. This is a tall order by any standard. However, as we daily face the unpleasant alternative, the need to put the car in its rightful place – using it where it is the best option and limiting it elsewhere – becomes increasingly clear. Building a new urban transport system, as opposed to making small changes to existing dysfunctional systems, will be a long-term undertaking. Therefore, we had best start soon.

35. The information industry

Braden R. Allenby

INTRODUCTION

The expansion of the information industry and its underlying infrastructure has been an integral part of the evolution of the human species. From the initial development of verbal capabilities, and an oral history, to the invention of the printing press, to technologies such as the telegraph, radio and television, information infrastructure has been a critical enabler of cultural development and diffusion. Frequently, in fact, information technologies are an integral part of the technology cluster characterizing a particular era: for example, the telegraph and railroad evolved together (Grübler 1998, p. 212). The evolution of modern mass media, combined with the growing power of the Internet, however, has created a new and powerful technological and cultural dynamic. Scholarship about the former is relatively advanced, but that is less true of the Internet. Like the global transport network that evolved as European civilization spread around the globe, the Internet will have huge impacts on the way humans perceive and manage the physical and biological world – and the way that world, increasingly endogenous to human society, evolves. It is thus a critical area of study for industrial ecology, the more so as it is so poorly understood at present.

Understanding the industrial ecology of the information industry is challenging for several reasons. First, the industry itself is in a period of rapid economic and technological change, with sector boundaries and core technologies undergoing fundamental transitions. Thus, for example, traditional telephone companies are merging with cable television firms to create facility-based broadband companies, while the underlying technologies shift from telephony to Internet protocol and wireless systems. Video and audio entertainment, which used to be provided on storage media (tape, CDs), in specialized locations (for example, movie theaters), or in broadcast mode (for example, over-the-air or coaxial cable-based networks) is now increasingly available in all forms over the Internet on demand. Additionally, many of the environmental implications of information industry evolution arise, not from physical platforms and facilities, but from the services provided: telework and e-commerce, for example, can be anticipated to have far greater environmental effects than those created by the underlying physical platforms. These 'direct' services are, moreover, dwarfed by the implications of building intelligence into the operations of virtually all sectors: 'smart agriculture' where sensors determine nutrient and moisture status of fields in real time; modern cars which depend on complex information systems to optimize operations across regional-scale geographic transport networks; linked networks of appliances which can be operated remotely as needed. Finally, there is the entertainment and content industry ('content' denoting these who create music, films, books and so forth), with profound but subtle effects on environmentalism and the cultural constructs, such as ‘wilderness’ and ‘nature’, that underlie it (Cronon 1996).

Many of these dimensions of the information industry are either beyond the core focus of industrial ecology or so poorly understood at this point that explication would be superficial. It is possible, however, to divide this industry into four components: (a) artifacts and physical platforms for services, such as switches, cable, computers and telephones; (b) operation of the networks and supporting infrastructure by facilities-based communications providers; (c) services which are built on these artifactual and network platforms; and (d) information provided over these systems and via these services. The industrial ecology dimensions of the latter are not, however, well studied or understood at all, and might well be considered by many to lie completely outside the proper purview of industrial ecology. Accordingly, this chapter will focus on three dimensions of the industrial ecology of the electronics and information industry: electronic products and design for environment (DFE); operation of physical communications networks; and services, with a focus on telework and e-commerce. The reader should always bear in mind, however, that electronics and information systems, especially at the current rapid pace of evolution, create not just incremental, but transformative, shifts throughout the economy, and that the most important of these may not be foreseen until they actually occur. Moreover, given the complexity and diversity of the systems involved, not all technologies can be dealt with in this chapter (thus, for example, we will not explore the industrial ecology of radio and television broadcast systems, or of wireless networks). There are two other critical information systems which are beyond the scope of this chapter. One is the market which, combined with the commoditization of virtually all values, is an important, decentralized information system (albeit incomplete, owing to externalities). The second consists of the genetic and higher-level information structures which are the province of biotechnology, genomic engineering and, at higher levels of the system, organismic studies, ecology and population biology.

DESIGN FOR ENVIRONMENT

Design for environment (DFE) is a now widely used term in industrial ecology, but it was first introduced in the electronics industry in the early 1990s (Allenby 1991a, 1991b; Sekutowski 1991; Allenby 1992a). In general, it was seen at the time as providing a mechanism for driving environmental considerations and constraints into the product design process (Allenby 1991b). More specifically, DFE was originally intended to be a subset of the existing product realization process then in use in the electronics industry, known as ‘Design for X’ or DFX, where the ‘X’ stands for a desired product characteristic such as testability, safety, manufacturability or reliability (Gatenby and Foo 1990). Although reflecting a number of then-current initiatives such as ‘green engineering’ or life cycle assessment (SETAC 1991; see also Chapters 12 and 28), DFE thus began as a module for a specific design process, and was only later broadened to become a generic set of practices capturing the application of industrial ecology principles to the design of complex products such as computers or telephone switches.

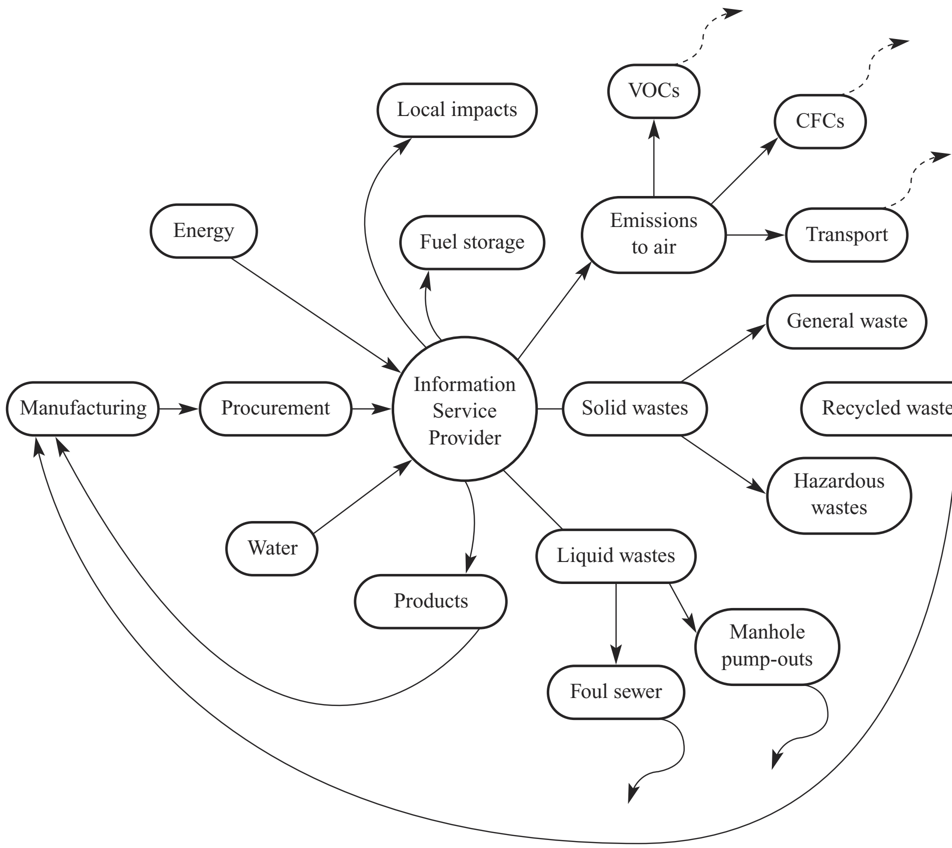
An important step in this evolution occurred with the formation of the Design for Environment Task Force at the American Electronics Association (AEA) in 1990, chaired by the author. This task force became the initial focus of developing DFE as an operative methodology for the electronics industry. An important step in this process was the production of 10 'AEA DFE White Papers' which introduced the broader electronics industry to the concepts of DFE. These white papers covered everything from materials recyclability to Design for Refurbishment to Design for Disassembly and Recyclability, and were subsequently issued as a package by the AEA (1993).[1] Thus was laid the conceptual basis for much subsequent implementation of industrial ecology principles in electronics manufacturing and product design activities. Equally important, perhaps, representatives of the US Environmental Protection Agency were invited to attend the AEA DFE Task Force meetings, which led directly to the subsequent formation of a DFE group at the Agency.

Another important early effort that significantly mobilized industry resources was supported by the Microelectronics and Computer Technology Corporation (MCC) in Austin, Texas, which from 1993 to 1996 issued a series of reports and roadmaps that explored in depth the material, energy, environmental and management issues raised by electronics products and components (for example, integrated circuit packaging, printed wiring boards, displays) over their life cycles (MCC 1993, 1994; 1996). The broad industry participation, from component manufacturers to assembly manufacturers to artifact producers, and the fact that those involved were primarily involved in research and development and manufacturing engineering, rather than in environmental and compliance activities, made these documents both technically rigorous and a significant departure from initial practice, which had too often seen environmental professionals trying to prescribe to design and manufacturing teams. From that point, the principle institutional DFE research initiative passed to the Institute of Electrical and Electronics Engineers, Inc (the IEEE), which has held annual symposia on electronics and the environment, focusing on DFE, from 1993 to the present. The proceedings from these symposia are perhaps the single greatest resource for exploring the progress and elements of DFE as it has evolved for the electronics industry (IEEE 1993–2000).

From about 1995 on, however, an important jurisdictional shift gradually occurred. While initially organizations centered in the USA, such as AEA and MCC, had led in research activities, the growing European interest in product takeback (whereby the manufacturer takes its product back after the consumer is through with it, and manages the recycling or disposal process), especially as applied to electronics, led to European research initiatives becoming dominant (for example, Clegg and Williams 1994; European Commission Project Group 1995; Hedermalm *et al.* 1995; Taberman *et al.* 1995; Ecocycle Commission 1996). Response by electronics manufacturers was global, but there was little question that the primary driver of technological evolution shifted from US waste reduction and remediation policies to European product takeback and life cycle product management initiatives (see Chapters 33 and 42).

THE INDUSTRIAL ECOLOGY OF INFRASTRUCTURE OPERATIONS

Direct infrastructure impact at the operation level is multidimensional in practice (Figure 35.1). Operating such an infrastructure involves many separate components and platforms,



Source: Based on data from BT Annual Report, *www.bt.com/corinfo/enviro/fact/index.htm*.

Figure 35.1 Information service provider environmental life cycle

from 'traditional' activities such as office buildings and vehicle fleets, to more specialized requirements such as extensive battery back-up systems for switching centers to provide service continuity if the electric grid goes down. While most firms face strong financial incentives to optimize elements of their infrastructure activities, such as energy consumption resulting from network operation, there are few studies of information infrastructure as a whole. Perhaps the best source is British Telecom's 1996–7 annual environment, health and safety report (BT 2000). The data in this report support certain conclusions about the relative importance of certain infrastructure activities compared to others. For example, it is fairly evident from the data provided that network and infrastructure operations, as opposed to office building and fleet operation, constitute the single biggest source of emissions (somewhere around 90 per cent of carbon dioxide and 98 per cent of sulfur dioxide – 1996–7 data). Most batteries recycled are lead acid network exchange batteries (99 per cent in 1996, 87 per cent in 1995 and 93 per cent in 1994), as opposed to vehicular lead acid or NiCad batteries. The major metal-containing residual stream from BT operations is from network switching exchange construction and replacement activities, as opposed to, for example, battery or cable recycling activities (averaging around 65 per cent over the period 1991–6).

Such systematic infrastructure data are obviously quite useful to the industrial ecologist: they help define the system quantitatively, identify major as opposed to minor material flows (and thus potential impacts) and point towards priorities for management of environmental impacts. It must be remembered, however, that this is the only study of its kind of which the author is aware. Moreover, these data are preliminary, not peer reviewed, and somewhat ad hoc, and reflect the operation of a particular kind of infrastructure (primarily telephony) at a particular point in technological evolution in a developed economy. Thus, for example, one would expect scrap rates for many developed country telephony systems to be reaching a peak as telephone switches are replaced by Internet protocol routers, and the assumption that virtually all lead acid batteries are recycled, which is valid in countries such as the UK, may not hold in many developing countries. In addition, recent technological trends that could have a significant impact on certain material flows – such as the reductions in paper consumption resulting from electronic billing, e-mail, business-to-business e-commerce systems and intranet deployment of electronic business and management documents – rapidly date such studies. Accordingly, this must be regarded as an area where the industrial ecology of the system is poorly characterized and understood, and one that is thus ripe for continued research.

THE SERVICE DIMENSION

A signal limitation of traditional environmental studies and policy, and, indeed, of industrial ecology in its earliest phases, was the exclusion of services from consideration. The first major responses to this significant conceptual gap were studies of the 'functionality economy', where function rather than physical product is provided to the end customer (see generally Stahel 1994) and this handbook, and, more technically, the example of Xerox (Berko-Boateng *et al.* 1993; Azar *et al.* 1995). The second has been the beginning of an industrial ecology focus on services, including work on methodologies (Hopkins *et al.* 1994; Lober and Eisen 1995; Guile and Cohon 1997; Graedel 1998; Allenby 1999a), policy (Rejeski 1997; Allenby 1997) and specific sectors (Davies and Lowe 1999; Davies and Konisky 2000; Davies and Cahill, forthcoming). The industrial ecology of services is obviously a huge research area just beginning to be explored; we will focus on the subset of services involving the information industry for the rest of this chapter.

Initially, it must constantly be borne in mind that, although our understanding of the social and environmental dimensions of some service areas such as telework is improving, we are far from understanding the social, environmental and cultural impacts of information infrastructure, and the services it enables. In part, this is due to basic psychology: our inability to understand the future except in terms of the past, a limitation that becomes especially apparent during times of rapid technological evolution. This is particularly important in the information industry, where the rate of technological change is unprecedented. Most people are aware of Moore's Law, and, indeed, as it foretold semiconductor performance is doubling every 18 months; moreover computer performance per dollar is doubling every 21 months. But similar rates characterize higher (more complex) levels of the system as well. Maximum Internet trunk speed was doubling every 22 months, as was maximum router speed, until 1997, when the doubling period shrank to six months. Internet traffic growth doubled every 21 months from 1969 to 1982, but the doubling rate shrank to nine months between 1983 and 1997 – and it now doubles every six months (Roberts 2000). Financials show similar growth. According to an influential study commissioned by Cisco and conducted by the University of Texas, the US Internet economy generated $301bn in revenues in 1998, and it is growing at a compound rate of about 175 per cent per year (for comparison, the US GDP rate of growth is about 2.8 per cent). The number of Internet economy jobs grew from 1.6mn in 1998 to 2.3mn in 1999, and each Internet employee produced revenue of about $250 000 (compared to, for example, automotive industry averages of around $160 000) (Bodman 1999; *The Economist* 1999).

The rapid pace of technological evolution, combined with the significant cultural dimension involved in information production and consumption, makes generalizations both difficult and, at this point, somewhat superficial (although there have been a number of efforts to explore the linkages between information technologies and sustainability, particularly in the European Union: for example, Information Society Forum 1997, 1998; Forum Info 1998, 2000). Rather, at this point analysis of scenarios is a more appropriate way to identify issues, learn to pose the right questions – and to appreciate the magnitude of our ignorance.

For example, consider the implications for entertainment of the evolution of broadband Internet and cable service to the home, combined with ubiquitous wireless access from virtually anywhere. It is very likely that in the near future the result will be the provision of entertainment of all kinds over the Internet to virtually any desired location in real time. The increasing ubiquity of the Internet, and the increasing acceptance and technical viability of audio and video streaming technologies, will dramatically collapse demand for CDs and tapes. In short, the media storage component of the home entertainment industry will be dematerialized. Concomitantly, the packaging, physical transportation and built infrastructure necessary to support the current system of material-based media delivery will be reduced or eliminated as well. This would be an environmental benefit: quality of life, in the form of on-demand entertainment and information of any kind, would be enhanced, with far less environmental impact. On the other hand, the economic costs to existing businesses, share owners and employees are apparent. These transition costs are not unique to environmentally preferable technological evolution, but are characteristic of constantly changing capitalist economies in general. Nonetheless, they do raise the issue of equitable distribution of costs and benefits arising from implementation of environmentally preferable practices (and the more difficult issue of how social, economic and environmental risks, costs and benefits are to be compared with each other). Moreover, one can envision so-called ‘rebound’ effects as well: is it not possible, for example, that on-demand delivery of Internet-based entertainment and information to cars will simply make traffic jams more pleasant, thereby reducing incentives not to get caught in one? If nothing else, this scenario clearly illustrates the need for systemic, industrial ecology-style analyses of such developments if important dimensions are not to be overlooked. It also raises the more subtle question of where the bounds on an industrial ecology study of a service should be drawn: should it, for example, include the (reasonably) predictable economic and social effects as well?

Another familiar scenario is provided by telework. At many companies, telework is increasingly recognized as an important ‘triple bottom line’ technology. It benefits firms economically, because they save on rent and can retain valuable employees, and because teleworking employees are generally more productive. It provides social benefits because employees and their families enjoy a higher quality of life. Moreover, traffic congestion, a major problem in many urban and suburban areas, is reduced, which benefits everyone who uses the roads. It provides environmental benefits because emissions are reduced if unnecessary commuting is limited; moreover, to the extent that congestion is eased, emissions from all vehicles are reduced marginally as well (Allenby and Richards 1999; FIND/SVP 1997). The data underlying these conclusions are, however, sparse and incomplete to some degree, and the extent to which economic and cultural patterns will change over time in unpredictable ways must be considered in any comprehensive cost–benefit assessment. For example, an earth systems engineering and management (ESEM) approach might lead one to ask whether in the longer term the availability of Internet infrastructure, combined with the delinking of place from work, might not lead to completely different patterns of built environment with concomitant implications for demand for products (enhanced e-commerce), transport systems (more dispersed populations requiring greater private transport), impacts on the vitality of urban centers, or increased gaps in quality of life between the 'haves' (knowledge workers that could choose where and how to live, and pull entertainment, information and personal connectivity from the Internet) and 'have nots'. As with the first scenario, such questions raise difficult questions concerning the appropriate bounding of industrial ecology inquiries into information services.

The analytical difficulties become more apparent when one considers the much more complex scenario of e-commerce, which has exploded in recent years (Bodman 1999). Here, there are no analytical structures or methodologies by which to begin evaluating the environmental implications of such a complex phenomenon (Cohen 1999; Rejeski 1999). To begin with, e-commerce has very different characteristics depending on the agents involved. Generally, four categories are recognized: business-to-business (B2B), business-to-consumer (B2C), consumer-to-business (C2B) and consumer-to-consumer (C2C). Most people think of e-commerce in terms of on-line purchasing, or B2C, but this activity is fairly small compared to B2B: in 1998, for example, B2C e-commerce was worth perhaps \$8bn, while B2B e-commerce was around \$43bn (*The Economist* 1999). C2B and C2C, by contrast, are far smaller and less developed segments.

One can certainly make reasonable assumptions under which e-commerce offers the potential of significantly improving the environmental efficiency of the economy. Consider, for example, B2C. Each individual consumer does not have to drive their personal vehicle to a mall; rather, the fleets of mail and parcel delivery services run along their efficient, pre-planned routes. Is this environmentally good or bad overall? Probably good, but no one really knows: it depends in part on things like the transport modality – air, truck, rail – chosen for various stages of the product distribution system. It also depends on the efficiency of the routing algorithm used by the delivery service. In this regard, it could be said that the advances in mathematics that have enabled more efficient routing of vehicles among numerous points are possibly one of the most potent environmental technologies of the past decade. Additional complexities and potential reflexivities arise because 'softer', non-economic cultural patterns are also involved in this case: for example, is demand for e-commerce modulated by the attraction of social interaction at malls? Even if I buy my shirts over the Internet, will I still drive to the mall to hang out with my friends?

B2B systems, on the other hand, raise a completely different set of issues. For example, B2B practices can cut waste significantly, leading to dematerialization of existing industrial operations. B2B systems can enable companies to shift from offering products, to offering services, with concomitant dematerialization of economic activity. Consider Home Depot, which has shifted from simply selling materials to small contractors – its most important customer segment – to offering a web site service. Contractors log in, enter details of their job, and the Home Depot software calculates what they will need and arranges for just-in-time delivery to the job site, eliminating the usual industry practice of overestimating materials, which then get wasted. What is ordered gets used. Or consider the new practice of printing books on demand: each book has a customer waiting, eliminating huge amounts of paper and energy which would otherwise be used to print and distribute books which never get sold, and end up being returned to, and discarded by, the publisher. E-commerce can also be a powerful source of dematerializing demand for paper in other ways. AT&T estimates indicate, for example, that some 15 million pages of paper use per month have been eliminated by going to paperless, Internet-based, billing systems for AT&T customers. On the business-to-business side, AT&T will eliminate the use of over 1.5 million pages of paper per year by going to e-commerce links with just one major supplier.

Such dematerialization processes can be driven by strong economic forces. Consider the case of Dell Computers, for example. The usual practice in the computer industry is to maintain a 60–80-day average inventory. Especially given the rapid pace of technological evolution in that industry, that translates into a lot of inventory that never gets sold, thus becoming waste (and the warehouse space and transport systems that are used, as well as the manufacturing impacts embedded in the product, are also wasted). Dell, on the other hand, uses its e-commerce systems, both upstream and downstream, to operate on about six days of inventory. This e-commerce system has created a profound shift in the economics of Dell's operations: in 1990, Dell had sales of \$546mn and required \$126mn in net operating assets to run its business. By 1998, however, Dell had \$18.2bn in sales using only \$493mn in net operating assets. Operating assets as a percentage of sales declined from 23 per cent to 3 per cent, while return on invested capital was up from 36 per cent in 1990 to over 400 per cent in 1998 (Bodman 1999). In short, as in the case of energy consumption per unit of economic activity, anecdotal evidence suggests that the Internet could be a powerful dematerialization technology.

But, lacking industrial ecology-type studies of e-commerce, it is simply too early to tell whether these dematerializing and energy-efficiency effects are important or not. No one honestly knows – and, in truth, no one knows how to go about thinking about these issues yet. Some studies (for example, Romm 1999) have concluded that e-commerce has the potential for significant dematerialization and decarbonization of at least developed economies. These have been heavily criticized (for example, Lake 2000) and are, at a minimum, premature: for example, the Romm study overlooks the well-known phenomenon that microeconomic efficiencies tend to translate at the macroeconomic level into shifts in supply and demand curves that result in higher consumption, thus swamping any economy-wide environmental efficiency effects (Grübler 1998). The challenge is broader than simply gathering economic, energy or material consumption data, however. Not surprisingly, the cultural dimensions of information systems may have profound (and subtle) effects on individual perception and cultural construction of critical concepts. For example, television and now the Internet are subtly changing the way that people understand and relate to concepts such as 'environment', 'nature' and 'wilderness'. One effect is to substitute consumption of information for experiencing nature first hand, of course, but it goes farther than that: rangers have for years noted that people raised on 'TV nature' tend to do things like try to hand-feed bears in national parks, with discouraging consequences (for the people). There is, in other words, a large and growing gap between their concepts of 'nature' and the real bears that maul them. In addition, global market forces combined with information systems and e-business encourage the tendency to commoditize nature, a process foreseen by, among others, Marx. How many people experience 'nature' in Nature Company and other upscale stores in fancy malls, or are exposed to 'wild animals' in parks and 'sea worlds', carefully designed by corporate sponsors to present a particular concept of wildness (Cronon 1996; Harvey 1996)? It is a reasonable hypothesis that, just as many people had a very different concept of 'wilderness' or 'nature' a century ago than we do now, future generations will also attach far different meanings, and contexts, to such terms – and the Internet, and associated information systems such as virtual reality products, will undoubtedly have major impacts on these cultural developments.

NOTE

1. The ten white papers were (published as AEA 1993): 'What is Design for Environment?' (Braden R. Allenby, AT&T), 'DFE and Pollution Prevention' (Braden R. Allenby, AT&T), 'Design for Disassembly and Recyclability' (R. Grossman, IBM), 'Design for Environmentally Sound Processing' (J. Sekutowski, AT&T), 'Design for Materials Recyclability' (W. Rosenberg, COMPAQ, and B. Terry, Pitney-Bowes), 'Cultural and Organizational Issues Related to DFE' (Braden R. Allenby, AT&T), 'Design for Maintainability' (E. Morehouse, Jr, USAF), 'Design for Environmentally Responsible Packaging' (K. Rasmussen, General Electric), 'Design for Refurbishment' (J. Azar, Xerox) and 'Sustainable Development, Industrial Ecology, and Design for Environment' (Braden R. Allenby, AT&T).

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PART VI

Applications and Policy Implications

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36. Industrial ecology and green design

Chris T. Hendrickson, Arpad Horvath, Lester B. Lave and Francis C. McMichael

In essence, engineering is concerned with the creation of products and processes. Engineers need to know how to make things with desired properties. The traditional properties of interest have included performance, reliability and cost. Reducing the environmental impact of new products and processes is of increasing importance. Increasingly stringent regulation, widespread public concern and new corporate environmental policies motivate this environmental interest. Of special importance for implementing environmentally conscious, 'green' design are appropriate knowledge, tools, production methods and incentives that can be applied during the design process.

There is a growing body of knowledge in the form of design heuristics and tools for green design. With its focus of system-wide resource flows, industrial ecology is a major source of such approaches. However, economic and engineering issues predominate in green design. Moreover, many methods used for environmental analysis are not appropriate for design decision making. Ecological inventories and conventional product life cycle assessment operate with data needs and time requirements that make them difficult to incorporate into the design process, especially during the important conceptual design stage. Even conventional tools such as stochastic modeling and decision analysis can inhibit the conceptual design phase or vastly lengthen the time taken to consider each alternative. Ideally, green design tools would be 'invisible' to the designer save as additional attributes to consider in evaluating a design. We must select the places to intervene carefully so as to reduce complexity and be certain that the new tools and environmentally conscious objectives are appropriate and useful. In particular, designers need useful environmental standards representations, access to the latest technology opportunities, easily applied rating systems for design impact on the environment, and tools to help assess life cycle implications, including accounting systems to reflect the full environmental costs of new designs.

GREEN DESIGN

Decision making during the design process is critical for the achievement of environmental goals. It has been estimated that 70 per cent or more of the life cycle cost of a typical product is determined during design (USNRC 1991). Much of the production process and operational impacts of a product are determined through decisions such as material choices or component power consumption. A traditional means of achieving environmental requirements is to add 'end-of-pipe' control processes for waste treatment. While a large proportion of environmental engineering still focuses on waste treatment processes, it has become increasingly clear that pollution prevention and waste reduction through design can be much more effective (Bishop 2000; Fiksel 1996; Reijnders 1996; USOTA 1992).

How can we effectively influence design decisions to achieve environmental goals? Design is a complicated and rather messy process, with numerous competing requirements and a premium on rapid decision making. Designers tend to be overloaded with conflicting criteria and information. Aids for green design must be easily and rapidly applied to be effective. Assessing environmental impacts requires analysis of downstream effects during production, operation and disposal. We especially wish to identify design changes that have low cost but considerable downstream benefits. For example, we may wish to include features such as 'snap fits' that have little or no extra initial cost but make disassembly and thus re-use easier (Kirby and Wadehra 1993). Effectively incorporating environmental concerns in decision making requires extension or modification of synthesis methods to consider a wider range of alternatives and requirements. For example, applicable environmental standards must be available in a relevant form to assess a design's acceptability or to suggest appropriate modifications. Figure 36.1 illustrates the product life cycle from conceptual design through to disposal and other end-of-life options.

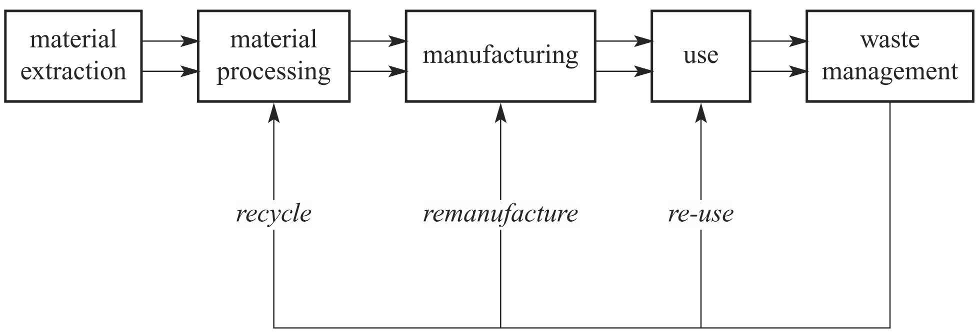


Figure 36.1 Product life cycle

Design processes vary tremendously in extent, duration and complexity. Typical dimensions of variation include the following:

- the extent of innovation sought;
- modularity of the problem to permit decomposition;
- the degree of regulatory constraints;
- the type of information flow in the design process;
- the magnitude of the design activity;
- the diversity of specialization domains involved;
- the magnitude of the designed product or process;
- the extent to which geometry is important;
- the number of products or processes to be produced; and
- longevity of the resulting product or process.

Different strategies for information dissemination, incentive structures and design aids may be required along these different dimensions, but there should be a core concern for environmental objectives related to toxic materials discharges, energy use and material use.

Green design often relies on practical knowledge (Burall 1991; Fiksel 1996; Reijnders 1996). Green design methods typically take the form of design heuristics, technology examples (such as solvent substitutions) and waste re-use examples (for example, waste heat productively sold). Few automated design aids are available, while many design processes rely on computer aids for geometric representation, tolerancing and synthesis.

Defining appropriate objectives for green design is a contentious issue in its own right (Lave *et al.* 1994). Maximizing recycling is not appropriate since the recycling may occur at inordinate cost or incidental environmental harm. Similarly, eliminating all toxic materials in products is not appropriate since the toxic materials may be useful and their discharge into the environment prevented during and after use. Following an industrial ecology perspective, we will focus upon toxic material discharges to the environment, energy use, and material use and re-use in what follows.

What has been driving design decisions towards more environmentally conscious products and processes? Without being comprehensive, some important incentives can be noted. Direct regulation of technology and emissions has had an undeniable impact, but has also resulted in expensive adversarial relationships. Recognition of eventual liabilities for product and facility disposal has occurred, although the extent to which such remediation liabilities are abstracted to the design stage is questionable in many cases. Opportunities to gain by marketing 'green products' (or restricting sales of competing, non-green products) also exist, especially for leasing or taking back products and then handling recycling in a closed loop. Technological changes also create opportunities, as in CFC substitutes or new energy-efficient very large-scale integrated (VLSI) chips. Proactive changes such as reductions in toxic releases are also apparent in many companies. Reporting programs have focused more attention on environmental performance by managers, the public and investors. Finally, voluntary standards such as the ISO 14000 Environmental Performance Standards have induced change in participating organizations (Cascio 1996; ISO 1998).

In the next sections, we present three examples of green design: computers, cars and structures. In these examples, we emphasize particular decisions and impacts for illustration, especially energy use (associated with the use of non-renewable resources), material selection and toxic material discharges. In a concluding section, we return to the issue of what changes are needed in public policy and design methods to capture these opportunities.

EXAMPLE: GREEN CAR DESIGN

Green car design can affect the volume and kind of waste materials, material re-use, energy consumption and environmental discharges. Since cars are large material sinks and consume large amounts of non-renewable petroleum products, green design is important. Cars require roughly two-thirds of the iron, half the rubber and one-fifth of the aluminum produced in the USA per year. Over 470 billion liters of petroleum are consumed in motor vehicle operation annually in the USA. While US motor vehicle use is still a significant fraction of world production, it is declining as production increases in other parts

Table 36.1 World motor vehicle production (thousands of vehicles)

Year	USA	Canada	Europe	Japan	Other	World total	USA % of world total
1997	12119	2571	17773	10975	10024	53463	22.7
1996	11799	2397	17550	10346	9241	51332	23.0
1995	11985	2408	17045	10196	8349	49983	24.0
1994	12263	2321	16195	10554	8167	49500	24.8
1993	10898	2246	15208	11228	7205	46785	23.3
1992	9729	1961	17628	12499	6269	48088	20.2
1991	8811	1888	17804	13245	5180	46928	18.8
1990	9783	1928	18866	13487	4496	48554	20.1
1985	11653	1933	16113	12271	2939	44909	25.9
1980	8010	1324	15496	11043	2692	38565	20.8
1970	8284	1160	13049	5289	1637	29419	28.2
1960	7905	398	6837	482	866	16488	47.9
1950	8006	388	1991	32	160	10577	75.7

Source: American Motor Vehicle Manufacturers Association

of the world (Table 36.1). The car also creates system-wide demands for greater roadway infrastructure and travel (Graedel and Allenby 1998).

One positive aspect to car use is the extent to which material recycling is being achieved. Recycling car components and materials is a large industry, with some 10 million scrap vehicles processed annually (AAMA 1993). In the disposal process, valuable components are first removed for either direct re-use or special disposal. The remaining hulks are usually shredded and metal pieces recovered. In Europe, there are some initiatives among motor vehicle manufacturers to take back old vehicles and recycle a large fraction of the vehicle mass, so even more of the vehicle may be re-used.

Non-metallic waste from the shredding operation (known as ‘fluff’) is generally not recycled (Isaacs and Gupta 1997). It is made up of plastics, fluids, dirt, rubber, fabric, glass and other materials, commonly referred to as the car shredder residue. While fluff is generally treated as a non-hazardous waste, potential troubles exist: California considers it a hazardous waste, while Rhode Island and Massachusetts require testing for hazardous content (Dvorak 1993). Automotive fuel tanks have been made from steel for many years, but plastic tanks have recently been introduced to save on vehicle weight (Joshi 1999). The advantages of molding complex shapes allows easier placement of the fuel tank in a vehicle. Complex geometries make the tank harder to drain at the end of life. Gasoline absorbed into the plastic that may be released in a drained tank creates problems for shredders. Plastic tanks with combustible mixtures are a hazard for shredding equipment.

Engine emissions from car use are subject to increasingly stringent regulation. There is growing interest in moving from ‘end-of-the-pipe’ treatment systems (such as catalytic converters) to alternative fuels and hybrid and/or electric cars. A difficulty with electric battery cars is the toxic materials in the batteries, especially over the life cycle of battery recycling, even if only traces are lost to the environment (Lave *et al*. 1995; McMichael and Hendrickson 1998).

Toxic Discharges into the Environment

Motor cars contain numerous toxic materials, usually in trace amounts, such as cadmium in tires or in paints or lead in solder. Because of the shredding process, trace toxic materials in car hulks can be a problem since they may make the shredder fluff hazardous. Several motor vehicle manufacturers have begun to phase out toxic materials such as cadmium completely. Again, reducing the fluff toxicity depends upon design decisions. More complete disassembly of vehicles and component re-use provide an alternative disposal method for some or all of the toxic content (Isaacs and Gupta1997). For example, car bumpers are now disassembled and can be re-used in parts for new vehicles.

Lead acid starting–lighting–ignition (SLI) batteries have received special attention. Where recycling is almost universal, batteries constitute an environmental hazard only to the extent that accidental spills occur or battery returns are avoided illegally. The overall use of lead has also been reduced in the USA. A typical 1974 battery contained about 12.5kg of lead, whereas a 1994 battery contained only 9kg. With 80 million SLI batteries sold annually in the USA, this weight reduction reduced overall lead use by 280 thousand tons (280kMT) (USOTA 1992). Next-generation vehicles call for increased electrical or non-motive power for conventional vehicles. Higher electrical power demands are likely to result in changes in the system voltage from the universal 14 volt system to either a 42 volt or a dual 14/42 volt system (Kassakian *et al.* 1996). These higher volt systems will include increases in the number or size of batteries in conventionally fueled vehicles. Battery issues are not limited to hybrid or electric vehicles, but remain a continuing concern as the demand for more electrical and electronic components are added to vehicles.

As in most manufacturing processes, there is a variety of toxic materials used in production and assembly of motor vehicles. As in most processes, substitutes are often available. For example, using water-based paints eliminates the use of organic solvents in paint.

Material Use and Re-use

Designers select about 600 materials for a modern car. For example, a 1990 Ford had 47 different plastics in the dashboard. Designers have found that plastics have many desirable attributes, such as reduced weight to boost fuel economy, leading to replacement of much of the steel used in earlier models (Curlee *et al.* 1994).

The proliferation of materials impedes recycling. Car shredding as currently practiced is intended to recover steel. With some additional effort, other metals can be recovered. Relatively little attention has gone into recycling plastics, however. Thus the increased substitution of plastic for steel has squeezed shredder operators economically, decreasing their revenue from shredding a car and increasing their costs (by giving them more fluff to dispose of).

Recycling aluminum is particularly desirable, since a great deal of energy is required to make virgin aluminum and little is required for recycling. Roughly 60 per cent of the aluminum used in cars is recycled (AAMA 1993), which saves 90 per cent of the energy required to smelt virgin aluminum. Nonetheless, both plastics and aluminum suffer from the problem that recycling mixed types (either different alloys or different resins) lowers the quality of the recycled material relative to the virgin material (Isaacs and Gupta 1997).

Energy Use

Cars are prodigious consumers of energy, especially in the form of non-renewable petroleum. Operating energy consumption dwarfs manufacturing and recycling energy costs and savings. Vehicle recycling represents net energy savings since recycled metals require less energy than in production from ore, but weight reduction would save more. Particularly as a result of greater fuel efficiency, life cycle energy use is declining. From 1980 to 1990, the overall life cycle energy consumption of an average car declined from 688 to 590 million BTU, a decline of 14 per cent. The average fuel efficiency over this same period went from 11.5 to 9.8 liters per 100 kilometers, an improvement of 18 per cent (Curlee *et al.* 1994) and has improved somewhat more in recent years.

Improvements in fuel efficiency have been achieved primarily by 'downsizing', power train improvements, reduction in rolling and wind resistance and some dematerialization. More recently, lighter materials, such as aluminum or composites, have been gaining greater use. Audi has introduced an all-aluminum car model. The entire design process must be involved to make such changes effective, since changing one component will affect others. See, for example, Lovins (1996) for a discussion of 'hypercars'. Over an entire lifetime, a weight reduction of 100 kg will reduce fuel consumption by roughly 736 liters of gasoline (Sanders and Wood 1993). All of these improvements stem from design changes and market demands, with some regulatory push due to corporate fuel economy constraints on manufacturers.

Existing technology could achieve higher fuel economy, thereby reducing petroleum demand. Without substantially higher gasoline prices, there is little market incentive to introduce more energy-efficient vehicles that are smaller, less safe or more expensive. It remains to be seen if new technologies can achieve fuel efficiencies without sacrificing other attributes. For example, Table 36.2 reports estimates of the relative advantages of

Table 36.2 Evaluation of attributes for fuel–engine combinations relative to a conventional car

	Reformulated gasoline	Reformulated diesel	Compressed natural gas	Ethanol from biomass	Battery electric vehicles	Hybrid electric vehicles	Fuel cell
Vehicle emissions							
A-I: ozone, NOx, VOC	2	−1	3	2	4	3	3
A-II: particulate matter	*	−1	*	*	2	1	1
A-III: air toxics	1	−3	2	1	3	3	3
Fuel-related							
B-I: fuel cycle emissions	−1	−1	1	*	−1	*	*
B-II: fuel cost	−1	1	1	−4	1	*	*
Vehicle performance							
C-I: range	*	1	−2	−1	−4	1	1
C-II: vehicle cost	*	*	−1	*	−2	−2	−2
Social issues							
D-I: infrastructure cost	−1	−1	−2	−3	−1	*	*
D-II: energy independence	−1	1	1	4	1	1	1
D-III: global warming	−1	2	1	5	*	2	2
D-IV: fossil fuel depletion	−1	1	*	5	1	1	1

Note: Baseline: unleaded, non-oxygenated, non-reformulated gasoline; air emissions compliant, tier 1 motor car (−5 worst,* comparable, +5 best).

several different new fuel–engine combinations relative to conventional cars (Lave *et al.* 2000). None of the new alternatives is best on each of the different impact measures, so the design problem is one of making trade-offs among desirable characteristics.

EXAMPLE: GREEN COMPUTER DESIGN

Computers are in a period of rapid technological change, with much computer hardware becoming obsolete within a few years. The worldwide number of personal computers is expanding rapidly. Of the computers discarded every year, many are functional, but obsolete. The rapid introduction of new technology results in heavy material demands. Environmentally conscious computer design is a continuing concern (Anzovin 1993).

Toxic Material Discharges

The manufacturing process in the electronics industry is a significant source of toxic materials. Toxic chemicals are used in a number of applications, such as circuit board cleaners, solvents and material components. Regulation, disposal costs and internal corporate policies provide strong incentives to reduce discharges of toxic materials. For example, Intel Corp. paid $2000 to have a ton of toxic waste handled in 1990 (Damian 1991).

The US computer manufacturing industry has gone through a remarkable process of reducing chemical releases (MCC 1993; Perry 1993). The 1987 total emissions of 14.7kMT were reduced by 69 per cent by 1990, to 4.6kMT (MCC 1993). Although the reduction affected some known carcinogens (for example, dichloromethane, 67 per cent reduction) and highly toxic chemicals (for example, hydrogen peroxide, 95 per cent reduction), the use of some other problematic compounds remained large, or even tripled in the period (for example, chromium compounds, another known carcinogen). The Environmental Protection Agency (EPA) has the 33/50 program to achieve a faster reduction of target chemicals such as mercury, cadmium, lead and so on. These reductions often require significant manufacturing process changes.

Toxic materials are present in different computer parts as well (mercury switches, plastics and so on). Monitor screens contain lead; they must be sent to a special site for disposal. In 1993 alone, an estimated 7.2 million monitors were scrapped (MCC 1993). Increasing the recyclability of cathode ray tube (CRT) components is desirable, and so is the alternative use of energy-saving and economical flat screen displays. Some plastics contain traces of toxic materials, such as cadmium, lead and chlorine. At least 85 per cent of reclaimed monitors are now recycled, substantially reducing landfill waste (Matthews *et al.* 1997).

There is a strong trend in the computer industry towards lighter and, in many cases, portable machines. These smaller machines have lower material demand, contain fewer toxic materials (except the batteries) and tend to be energy-efficient. Batteries are of particular concern because of their high toxic content, their relatively short lifetime and their tendency to be mixed with other solid waste. The heavy metal content of batteries is a significant contributor to the toxicity of municipal solid waste incinerators. Unlike the case of car lead acid batteries and recycling systems common in Western Europe, the USA does not have comprehensive systems in place for recycling computer batteries. Some computer manufacturers accept returned batteries for recycling, but the practice is not widespread. Lankey and McMichael (2000) compare the economic, resource and environmental impact for single-use and rechargeable batteries using a method based on Leontief input–output analysis. This work restates the benefits of reduced mass of waste batteries and reduced life cycle emissions from selecting rechargeable batteries over primary batteries whenever possible.

Improved process control and careful selection of materials in the design stage can improve current practice. Dematerialization and substitution of a less toxic material reduces the demand for hazardous chemicals and may save the company money (Damian 1991). If their use is necessary, toxic materials should be concentrated in removable subparts (for example, batteries) to facilitate recycling.

Material Use and Re-use

Design for disassembly and longevity may some day allow computers to be easily upgradable and refurbishable (Hendrickson *et al.* 1994). Some components of discarded computers are re-usable and contribute to the positive value of discarded computer equipment.

Chips and fans may live a second life in next-generation equipment, or in a product of less value (such as chips in home appliances or modern toys). It is anticipated that in the future cutting-edge electronics equipment may be leased to demanding companies, and be passed on to other customers after becoming obsolete and having gone through some refurbishing (Damian 1991).

Computer companies are switching to single-polymer and steel cases in their designs. Along with ‘snap-fits’, no glues, no paints, no composites and only a few fasteners, an environmentally conscious selection of materials facilitates less material use, easier disassembly and recycling.

EXAMPLE: STRUCTURES

The world started the 20th century with 1.6 billion inhabitants. The 6 billionth person was born in year 2000. Most of the inhabitants of our planet want the goods and services of the developed economies. The construction sector must provide the infrastructure for the world population. Owing to longevity, design changes take a long period to affect the overall stock of facilities. Constructed facilities have also seen much less design change due to environmental concerns than other products.

The construction industry is one of the most significant economic sectors (accounting for about 7.5 per cent of the US GDP in 1997 (US Statistical Abstract 1999) and, by virtue of its size, one of the largest polluters. Regulatory and public pressures on the sector are mounting. For example, the EPA has recently proposed lower emission standards for heavy diesel engines in construction machines (Phair 1998). Customers increasingly demand environmentally improved construction products and processes. In 1995, the US EPA issued a federal procurement guideline requiring agencies using federal funds to favor the purchase of cement and concrete products containing fly ash (USEPA 1995b). A proactive approach is needed in research and education, precursors for lowering environmental burdens while maintaining sustainable growth.

The common perception is that construction is using up non-renewable resources (energy, ores, ecological niches and so on) in a rapid and unsustainable fashion, and is polluting the environment disproportionately. For example, construction and demolition sites were labeled the most common source of industrial water pollution in England and Wales (UKNRA 1995). The industry is increasingly pressured by environmental stakeholders (public, government, academia and so on) to act in the interests of the environment, to remediate its contaminated sites, clean up production processes, reduce waste, use fewer materials and less energy, dispose of waste in a safe way, reduce hazardous emissions, re-use end-of-life products, recycle and so forth. Yet the construction industry has been lagging in environmental efforts such as environmentally conscious materials selection, design for disassembly, environmental performance measurement, process redesign, material re-use and recycling and product takeback by producers after use. In contrast, the electronics and office equipment producers, the chemical and the automotive industries have been at the forefront of environmental efforts (for example, MCC 1993; Graedel and Allenby 1995, 1998) and claim to have reduced environmental burdens while saving money on process optimization.

While several environmental issues have already been studied in construction (Moavenzadeh 1994), such as ecological impacts of new facilities (Conrad 1995), energy use of buildings (Bevington and Rosenfeld 1990), embedded energy of construction materials (Boustead and Hancock 1979), selection of environmentally preferable alternative materials for constructed facilities (AIA 1997; Horvath 1997), indoor air quality, asbestos removal, radon in buildings, recycling of building materials (Donovan 1991; Baccini and Brunner 1991; Perez 1994; Bossink and Brouwers 1996), lead-based paint (Gooch 1993; Synder and Bendersky 1983) and air pollution due to traffic (Barth and Tadi 1996). Many others have not yet been systematically and quantitatively examined. As a result, the construction industry is generally viewed today as being less environmentally conscious than other sectors of the economy. Construction professionals need quantitative results, reliable analysis methods and specific guidelines for their work.

The drivers for environmentally improved construction engineering and management are becoming stronger. For example, the 1995 US Federal Procurement Guidelines (USEPA 1995b) require contractors to use sustainable practices if they wish to sell products and services to the government. Federal contractors must comply with the US Emergency Planning and Community Right-to-Know Act's (EPCRA) Toxics Release Inventory (TRI 1995) reporting requirement (USEPA 1995a), which forces them to account for some of their toxic chemical emissions. (Non-compliance with the EPCRA provisions can lead to contract termination.) Section 613 of the 1990 US Clean Air Act requires each federal department and agency to maximize the substitution of safe alternatives for ozone-depleting substances, and certify to the Office of Management and Budget (OMB) that procurement regulations have been modified appropriately (*http://www.epa.gov/ozone/title6/procu/procu.html*; accessed 15 July 1999). Recycling was included in the 1991 Intermodal Surface Transportation Efficiency Act (ISTEA). Section 1038 of the legislation stipulated that 5 per cent of roads built with federal funds in 1994 must use a pavement made from recycled tires. The percentage of roads using recycled tires should have grown to 20 per cent by 1997. This mandate has since been revoked owing – in part – to incidences of burning recycled tire embankments and leaching of tire crumbs in pavements. The recently published ISO 14000 environmental management standards are expected to be adopted by companies that want to do business internationally (ISO 1998).

Greater research efforts in the area of environmentally conscious construction are also appearing. EPA's Air Pollution Prevention and Control Division (APPCD), the Civil Engineering Research Labs (USACERL) and the National Institute of Standards and Technology (NIST) have formed a partnership to conduct research on the life cycle environmental effects of facilities, with the focus on the environmental evaluation of building materials. The Center for Construction and Environment at the University of Florida performs research in a variety of areas, including green construction materials and deconstruction. Overseas, the Building Research Establishment (BRE) in the UK has teamed up with 18 industry partners 'to create an agreed method to identify the environmental effects of building materials throughout their life cycle' (*http://www.bre.co.uk/bre/newslets/BRE_news/apr 96/news 4_96.html*; accessed 15 July 1999). The partners are from trade organizations representing manufacturers of construction materials and products such as aggregates, clay pipes, timber, PVC, cement and metals. BRE is also leading a project on the development of life cycle environmental assessment methodologies for building materials and components funded by the European Commission (BRITE EURAM III).

SOME OBSERVATIONS

This review of changes in the design of different artifacts suggests a few guidelines for achieving environmental goals.

1. New products and processes can change rapidly in response to changing incentives related to market demands or regulation. In many cases, existing technology can be deployed or modified to achieve new requirements. For example, existing technology can achieve significant automotive fuel economy savings, but the economic incentive to market fuel-efficient vehicles is missing.
2. There is still considerable scope available for low-cost design changes to achieve environmental goals. Better knowledge of design implications and new design tools can be effectively promoted to this end.
3. Computer aids to assess and help synthesize designs for easier disassembly, re-use and recycling may be useful to help designers. Variations in the extent to which environmental concerns are considered are apparent in comparing different industries or companies within industries.
4. There is a significant time lag in effecting change through design incentives owing to the lifetime of products and processes. Unfortunately, retrofitting existing products and processes is generally expensive.
5. Public policy to present incentives and constraints within industrial and design decision making should be considered with a clear eye towards the general impact. Recycling and reducing toxic materials are only strategies that may or may not effectively promote environmental goals, depending upon the circumstances.

37. Industrial ecology and risk analysis

Paul R. Kleindorfer

Risk analysis in industrial contexts consists of four integrated processes: (a) identifying underlying sources of risk; (b) determining the pathways by which such risks can materialize; (c) estimating the potential consequences of these risks under various scenarios; and (d) providing the means for mitigating and coping with these consequences. Specific risks, once identified, are usually characterized by the probability of their occurrence and the magnitude of their consequences, but many other attributes of risks may be of interest to individuals affected by these risks. Risks can have both positive and negative outcomes and can occur in any domain of a company's operations, from engineering to finance.

A great deal of work in corporate finance and insurance has gone into the design of efficient risk management instruments for risks that can be monetized (for example, Doherty 2000) and, to the extent that the consequences of these risks are borne by the owners of an enterprise, there are strong incentives for managers to make efficient choices in balancing risks and returns. This is not usually true for industrial risks having safety, health or environmental (SHE) impacts, since these impacts are often borne by the ecosystem and by uninvolved third parties, including future generations. Thus, for SHE risks, market forces are not usually sufficient to motivate a profit-oriented company to operate efficiently. Achieving efficient trade-offs here requires instead that industrial practice be tempered by regulation and public participation. Exactly how this should occur for various types of SHE risks has been a major area of development in the literature of industrial ecology and will be the focus of this chapter. It first considers the central drivers of risk analysis in industrial contexts, since this has motivated much of the research in this area. Thereafter, the chapter briefly reviews key elements of current approaches to industrial ecology (IE) for SHE risks.

FACTORS UNDERLYING INDUSTRIAL RISK ANALYSIS

In the industrial ecology framework, each company has a special role as a steward of the environment and ecosystem within which it operates. Naturally, this role of product stewardship and environmental waste and risk management encompasses suppliers and customers just as 'extended value-chain analysis' encompassed suppliers and customers in the traditional supply chain improvement process. Indeed, it is useful to review the factors underlying the focus and need for industrial risk analysis in parallel with the extended supply chain. Figure 37.1 shows the leverage points for such risk analysis, beginning along the supply chain from virgin resources to suppliers, through product transformation to customers. Risk analysis has the task of identifying and quantifying risks along this extended supply chain and determining appropriate mitigation and response strategies.

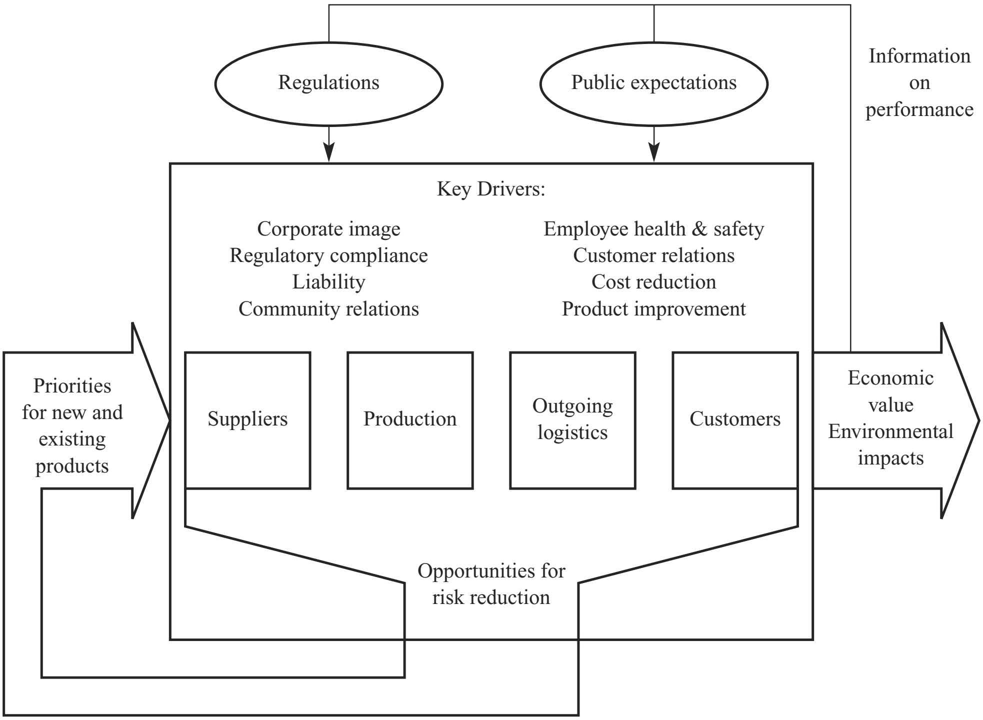


Figure 37.1 Risk analysis and the extended supply chain

The key processes of interest in the industrial risk analysis process are product and supply chain design to minimize SHE impacts; continuing waste minimization and risk mitigation after the product has been deployed; and diagnostic feedback from supply chain participants to assess opportunities for future risk reduction activities. Resulting SHE improvements in a company's supply chain can lead to economic benefits for companies in several areas.

Corporate Image

Mitigating SHE risks is not only socially responsible behavior; it is good business. First, owing to the public concern surrounding environmental issues, promoting environmental care can enhance a company's and an industry's image. The key driver from a business perspective is the avoidance of major accidents, which can have huge consequences for a company (see Klassen and McLaughlin 1996 for empirical estimates), as shown by the Union Carbide accident in Bhopal, India in 1984. As a result of this accident, Union Carbide itself lost its ability to operate as a company.

Regulatory Compliance

Regulatory compliance requires companies to track the use of hazardous substances and emissions of pollutants. While actual compliance clearly varies widely, especially among small firms with less to lose in the event of SHE incidents, major companies in the chemical and process industries have devoted significant resources over the past two decades to improving SHE performance (for example, Jaffé *et al.* 1995).

Liability and Negligence

Another factor driving companies to improve their environmental performance is the risk of being held liable, or found negligent, for accidents or environmental damage. This is true even when the company is acting prudently and using state-of-the-art technology. To limit liabilities, many companies implement strict risk reduction mechanisms, with a focus on generally lowering the level of pollution, biocides and toxics (P, B and T) associated with a company's supply chain and products produced (for example, Allenby and Richards 1994).

Community Relations

Improved relations with local communities and with other external stakeholders are becoming increasingly important for companies, as a matter both of law and of best practice (see, for example, McNulty *et al.* 1998).

Employee Health and Safety

Similar to community concerns, employee health and safety is a key focus of risk reduction and risk communication initiatives (CCPS 1989). Employee H&S is not limited to company workers or on-site exposures, but includes all parties in the supply chain who may be exposed to a company's product.

MANAGERIAL AND REGULATORY SYSTEMS TO PROMOTE EFFICIENT RISK MANAGEMENT

Integrating the supply chain to identify, mitigate and manage risks requires integration with key business processes, measurement of results and commitment from top management. A number of managerial concepts, tools and systems exist that promote these steps towards sustainable risk practices. Essentially, all the tools of industrial ecology noted throughout this handbook have important implications in promoting reduction of SHE risks. For example, life cycle analysis, gated DfX screens (where Design for X includes X factors such as Environment, Safety, Disassembly and Recycling) and reverse logistics all promote more sustainable products and supply chains and can have significant effects on the overall SHE risks of these supply chains by reducing their P, B and T content.

Of the new management systems to promote SHE risk reduction and sustainable industrial practices, the best known is certainly the Environmental Management System (EMS) under the international standards ISO 14000 (Carter 1999) and related systems such as those promulgated by the European Union under the Eco-Management Audit Scheme (EMAS). ISO 14000 began development in 1991, after the successful deployment of ISO 9000 standards, and the aspirations underlying ISO 14000 were motivated by the experience with ISO 9000. Recent empirical research shows that process mapping, as embodied in good IE practice and in the ISO 9000 quality standard, can be a significant aid to discovering process defects and fixing them (Angel 2000). By extension, this same logic of process excellence appears to apply to risks and ecological effects, and industrial practice is increasingly reflecting this belief (CCPS 1989; Bern 1998; Friedman 1997). Systems such as ISO 14000, EMAS and related process safety standards have become essential vehicles across the globe in codifying a company's SHE practices, in promoting auditing standards for these, and in providing the public and regulators with information about company SHE performance (Lofstedt *et al.* 2000; Kunreuther *et al.* 2000).

A prime mover for implementing IE is government regulation and economists have been concerned since the beginning of environmental regulation to provide the logical underpinnings for such regulation. In the context of the management of risk, economic models focus on the incentives a firm faces to engage in efficient risk mitigation. Theoretical and empirical results from this literature show that, if the firm does not internalize all the consequences of its decisions regarding risk, it will tend to underinvest in risk mitigation. This in turn awakens the call for regulation of such risks (see also Chapter 6). A number of models have been developed to investigate the consequences of various forms of regulation that attempt to rectify the noted underinvestment problem (for example, Shavell 1984; Gruenspecht and Lave 1989; Watabe 1999).

An alternative approach to risk regulation has been suggested by Kleindorfer and Orts (1998), and this approach is increasingly evident in North America and the European Union. The approach is called 'informational regulation' (IR) of risks and is based on the original ideas of Ronald Coase (Coase 1960). The IR approach requires firms to share information on their risk management programs and their performance with the public. The idea is to make it easy for NGOs, local community groups and affected citizens to identify the risks they face and, by providing thereby leverage for such groups, to put pressure on firms to improve their risk performance. Under IR, the locus of risk regulation moves away from government bureaucracy to local bargaining between affected communities, armed with legitimate information on the risks they face, and the companies giving rise to such risks. This approach is especially compelling when the risks involved are largely local and borne by an identified group of stakeholders, such as a community hosting an industrial facility.

THE TECHNICAL PROCESS OF RISK ANALYSIS

General procedures and a host of tools and industrial applications of risk analysis have been developed over the past half-century. A good survey of these is available in Haimes (1998) and further research progress is reported in the primary journal in the field, *Risk Analysis*. The accepted conceptual structure for managing SHE risks includes the following activities (see Friedman 1997):

- hazard identification: listing materials, processes and products of potential concern and qualitatively prioritizing these by their relative hazard;

- risk assessment: determining the credible releases, exposure pathways and events that might result from various events and scenarios and calculating the median and worst-case hazard zones associated with these;
- risk analysis: considering all safety systems, redundancies and mitigation possibilities, calculating detailed probability distributions for the hazards identified and considering damage reduction possibilities;
- risk management: specifying risk acceptance and risk reduction guidelines, specifying process hazards management procedures, including emergency response procedures, structuring financial and insurance provisions and establishing communication procedures with affected employees and the public.

For each of the above steps, companies can rely on a variety of tools and methodologies to assist them (Haimes 1998). In the hazard identification phase, data and procedures from various industry sources help to determine levels of hazard of various processes, materials and products. The tools of industrial ecology in mapping material and energy flows (for example, Ayres 1997b; Ayres and Ayres 1997) are essential diagnostic tools to indicate likely sources and magnitudes of emissions and wastes in key processes. In risk assessment, a variety of decision support systems are available for computing for various chemicals dispersion dynamics and their consequences. Risk analysis is supported by general tools, such as process simulation, decision analysis and event and fault tree methods, as well as by customization of these tools for use in specific industrial sectors. The literature on risk and insurance is well developed in the area of SHE risks (Freeman and Kunreuther 1997), as are approaches to emergency response. Finally, survey methods and other empirical approaches to measuring the perceptions of community members of the industrial risks they face are now widely developed (motivated by the work of Slovic 1987) to assist decision makers in determining what various stakeholders are concerned about and how best to communicate with them about risk reduction possibilities.

When these processes and tools are applied to a setting, the results achieved can be remarkable. In line with the findings of the effects of quality management programs, these results are in two key areas. The first is the structuring of a strong management system for companies that links their strategy process and their operations to a legitimate, science-based framework to identify, assess and manage their SHE risks. The second is an organic structure of shared knowledge that allows all stakeholders in a company's operations to understand the potential effects of these operations on the company's ecosystem. This systemic knowledge, and the purposeful activity triggered by it to achieve a sustainable fit between a company and its environment, is the most important characteristic of the industrial ecology approach to SHE risks.

RISK EPIDEMIOLOGY

The standard process of risk analysis, as described above, may use data from a number of sites, for example on the failure rates of specific fittings or equipment, but the results of the overall risk analysis process are usually quite site-specific. Recently, it has been suggested that standardized data from across a large number of sites could be used to provide more aggregate assessments of accident rates and severities. This approach, using data from a number of facilities, has been called accident (or risk) epidemiology, since it is similar to its medical cousin in attempting to use cohort cross-sectional or panel data to determine underlying factors of undesirable outcomes.

The best known example of this type of study is associated with the Risk Management Plan Rule under section 112(*r*) of the Clean Air Act Amendments in the USA, which is now briefly described. In the European Union, similar actions were undertaken under the Seveso II Directive (Orts and Deketelaere 2000). Section 112(*r*) sets forth a series of requirements aimed at preventing and minimizing the consequences associated with chemical accidental releases. These requirements are the basis of the US Environmental Protection Agency's rule on 'Risk Management Programs for Chemical Accidental Release Prevention' (hereafter the 'Rule'). The federal regulations promulgated under 112(*r*) apply to facilities (both public and private) that manufacture, process, use, store or otherwise handle regulated substances at or above specified threshold quantities (which range from 200 to 10000 kilograms). With some exceptions, the Rule requires all regulated facilities to prepare and execute a risk management program (RMP) which must describe the facility's approach to hazard assessments, worst-case scenarios and prevention programs. The Rule also specifies the requirement that regulated facilities maintain a five-year history of accidental releases and submit this history to the EPA no later than 21 June 1999, with further five-year filings expected in the future (probably repeated in 2004). The number of facilities that initially filed under the Rule was 14 500, with 1145 of these facilities (7.9 per cent) reporting 1913 accidents over the five-year period of interest. The database itself has been named RMP*Info and, except for sensitive data on worst-case scenarios, has been available to the public since August 1999.

The initial data were first analyzed in Kleindorfer *et al.* (2000). The basic approach followed was the epidemiologic methodology known as (retrospective) cohort study design. Epidemiology is the study of predictors and causes of illness in humans. Its use in studying industrial accidents had been proposed in a number of quarters (for example, Saari 1986; Rosenthal 1997). The motivating idea is to study the demographic and organizational factors of those facilities whose accident histories are captured in RMP*Info to determine whether any of these factors have significant statistical associations with reported accident outcomes, positive or negative, just as one might use demographic or lifestyle data for human populations to determine factors that might be associated with the origin and spread of specific illnesses.

To provide some sense of what accident epidemiology is, consider the basic demographics of the facilities that filed under RMP*Info. There are 14 500 facilities in RMP*Info and there are 1913 reported accidents in RMP*Info, with 1145 facilities reporting at least one accident. However, the sample size for various statistics will not remain constant at 14 500 and 1913, since some sites have multiple processes and some processes use multiple listed chemicals. For example, the average facility size among facilities reporting to RMP*Info, as measured in employee FTEs (full time equivalents), is 163 FTEs, ranging from facilities with less than 0.5 FTEs (recorded as 0 FTEs in RMP*Info) to 48000 FTEs. Half of the facilities have 11 FTEs or fewer.

Various accident rate statistics can be computed from the information in RMP*Info. For example, Table 37.1 reports accidents by listed chemical involved in the accident for chemicals involved in 10 or more accidents. The numbers of accidents by chemical ranged

*Table 37.1 Accidents reported in RMP*Info by chemical involved in the accident, 1994–9 (results for chemicals with 10 or more accidents during the five-year reporting period)*

Chemical name	Chemical ID	Number of accidents
Ammonia (anhydrous)	56	656
Chlorine	62	518
Hydrogen fluoride/hydrofluoric acid	55	101
Flammable mixture	155	99
Chlorine dioxide (chlorine oxide (ClO_2))	71	55
Propane	98	54
Sulfur dioxide (anhydrous)	49	48
Ammonia (conc. 20% or greater)	57	43
Hydrogen chloride (anhydrous) (hydrochloric acid)	54	32
Hydrogen	149	32
Methane	93	30
Butane	118	26
Ethylene oxide (oxirane)	9	19
Hydrogen sulfide	63	19
Formaldehyde (solution)	1	17
Isobutane (propane, 2-methyl)	107	17
Pentane	125	17
Titanium tetrachloride (titanium chloride ($TiCl_4$) (T-4)-)	51	15
Phosgene (carbonic dichloride)	10	12
Nitric acid (conc. 80% or greater)	58	12
Ethane	94	12
Oleum (fuming sulfuric acid)	69	11
Ethylene (ethene)	95	11
Vinyl chloride (ethene, chloro-)	101	11
Trichlorosilane (silane, trichloro-)	153	11
Methyl chloride (methane, chloro-)	5	10
Toluene diisocyanate (unspecified isomer)	77	10
Propylene (1-propene)	129	10

from 656 accidents for anhydrous ammonia facilities to a single accident for 22 listed chemicals. Exactly half of the 160 chemicals listed under the Rule were involved in at least one accident during the reporting period.

Consequences of accidents, according to various measures, are summarized in Table 37.2. We note in passing that there were no off-site deaths from the chemical industry in the five-year period in the USA. There were totals of 1897 injuries and 33 deaths to workers/employees, and there were 141 injuries and no deaths to non-employees, including public responders. Exactly half of the reported accidents (956 of the total of 1912) resulted in worker injuries. Table 37.2 also notes the damages to property and the non-medical off-site consequence analysis resulting from accidents during the reporting period. Note that the property damages alone are in excess of $1 billion, and they do not include business interruption costs, including losses in shareholder value and lost business

Table 37.2 Consequences of accidents during the reporting period (all 14 500 reporting facilities)

	Mean or total	Std dev'tion	Min	Max	Observations
On-site injuries to workers/contractors					
Total on-site injuries	1897				1912
Injuries per accident	0.9922	2.810	0	67	1912
Injuries per FTE per accident	0.0202	0.0784	0	1	1896
On-site deaths of workers/contractors					
Total on-site deaths	33				1911
Deaths per accident	0.0173	0.2224	0	6	1911
Deaths per FTE per accident	0.0003	0.0071	0	0.25	1895
On-site property damage ($ millions)					
Total on-site damage	1006				1907
Damage per accident	0.528	6.716	0	219	1907
Off-site property damage ($ millions)					
Total off-site damage	11				1907
Damage per accident	0.006	0.109	0	3.8	1907
Number of accidents with effects on the ecosystem					
Fish or animal kills	17				1913
Minor defoliation	54				1913
Water contamination	24				1913
Soil contamination	31				1913
Any environmental damage	101				1913

associated with accidents. These interruption costs are likely to have been significantly higher than the property losses.

In addition to the above descriptive statistics, a number of analytic studies on trends in the data, underlying risk factors and other multivariate analyses have been undertaken (see Kleindorfer *et al.* 2000). This research is concerned with characterizing the statistical association, if any, between such plant characteristics as size and accident frequency and severity, controlling for other demographic and process characteristics such as prevention level, the inherent hazard of the chemicals involved and other factors that might affect accident outcomes. Individual sector-specific and process-specific studies (for example, on chlorine plants) could also be coupled with other industrial ecology approaches such as material and mass balance studies. Using accident epidemiology to discover the organizational and process determinants of accidents seems a particularly fruitful approach given the growing availability of comparative, cross-industry data.

NEW FRONTIERS IN RISK ANALYSIS AND INDUSTRIAL ECOLOGY

The above areas represent the current state of risk analysis in respect to the industrial ecology movement. New frontiers include, first of all, the use of Internet technologies to improve shared knowledge about industrial impacts on the environment, globalization of activities related to risk assessment and management, and the continuing advancement of the science of risk analysis. In all these areas, the key element of risk analysis for industrial ecology remains linking of individual actors, be they government agencies, companies or households, to a science-based assessment of the impacts of their actions on their environment. The evolving paradigm of risk analysis continues to serve us well in this endeavor in determining and assessing sources of risk and in motivating the adoption of effective mitigation strategies to reduce these risks.

38. Industrial ecology and spatial planning

Clinton J. Andrews

This chapter links IE to geography and the planning of our built environment. First, it illustrates why spatial questions – 'where? how far?' – are important in IE. Second, it briefly discusses the difficulty and importance of aligning the IE perspective with political geography. Third, it defines spatial planning and describes its origins. The fourth and longest section shows how spatial planning practices affect the implementation of IE ideas. The chapter closes with an argument that there are important conceptual linkages between the intellectual projects of IE and planning.

Town planning, urban planning and regional planning are familiar, almost prosaic activities of governments worldwide. The connections to IE include both the practical, such as the design of industrial parks, and the conceptual, such as the role of utopian visions in guiding incremental decisions. Planners and industrial ecologists have much to offer one another.

WHY GEOGRAPHY MATTERS

Geography influences and even defines economic and ecological phenomena, and hence industrial ecologists cannot ignore it. Most obviously, resources are unevenly distributed in space, so that the bundle of environmental characteristics varies by location. Materials as diverse as air pollutants, water and mineral deposits all differ in concentration by orders of magnitude from place to place. The spatial incidence of humans and other species both reflects and influences this variation (Redman 1999).

Distance-related frictions affect the diffusion of species as well as the details of producer and consumer behavior, industrial location, market areas, innovation rates and settlement patterns (Thünen 1826; Weber 1929; Lösch 1954; Isard 1956; for a review, see Isard *et al.* 1998). Technological innovations such as motorized transport and the Internet have altered but not erased space as an economic variable (Blair 1995, pp. 60–65). The same is true of space as a biological variable, as invasive species problems attest (Sala *et al.* 2000).

By introducing geography we force questions about scale and level of analytical resolution into IE research and practice. But there is not a simple hierarchy from local to global; instead, there are multiple ways to delineate space – political jurisdictions, economic regions and ecological regions among others.

MISMATCHED BOUNDARIES

IE phenomena tend to obey economic or ecological boundaries rather than political boundaries; hence Socolow (1994, p. 12) has labeled IE as 'subversive'. River basins, which often cross political boundaries, are the focus of much work in IE (Russell *et al.* 1975; Ayres and Rod 1986; Stigliani *et al.* 1994; Brunner *et al.* 1994). The field has also held a long-term focus on global flows of materials (for example, Nriagu 1979; Thomas and Spiro 1994). However, industrial ecologists desiring to influence public policy must find ways to interest political decision makers whose concerns typically follow political boundaries (Bolin 1998). This task is easy when studies follow the coincident economic and political boundaries of the nation (for example, Lohm *et al.* 1994; Adriaanse *et al.* 1997), but it poses a significant challenge in small jurisdictions that are embedded within larger ecological, economic and political systems (Andrews 1999a).

Regionalism in the form of river basin commissions, air quality management districts and other regulatory entities matched to the scale of natural systems can be a solution to mismatched ecological and political boundaries. However, this is a politically weak solution because regional bodies exist at the pleasure of constitutionally defined units such as states or nations whose interests may not be congruent (Rabe 2000). Similarly, multilateral trade treaties and entities such as the North American Free Trade Agreement and the World Trade Organization attempt to match governance mechanisms to the scale of economic systems, with results that have limited scope and borderline political legitimacy (Vogel 2000).

Multi-level government is a structural approach to realigning varied phenomena with fixed political boundaries. Within a multi-level structure one can centralize some responsibilities in order to reduce interjurisdictional spillovers, ensure equity, enforce universal norms and pool risks; yet one can decentralize other responsibilities in order to allow experimentation, better match policies to local circumstances, reduce governmental power and preserve distinct civic cultures (Oates 1977; Elazar 1984; Osborne 1988). Multi-level government typically has four levels (nation, state province/region, county/district/department, municipality/commune) but it takes a variety of forms. The decentralized German and US federal systems divide sovereignty between the nation and the Länder/states, for example, whereas the centralized French and Italian unitary systems place the national government in a much more dominant position (Magstadt and Schotten 1988, p. 86). A fifth level of government has been added in Europe with the advent of the European Union. None of these forms resolves all jurisdictional mismatches, meaning that intergovernmental relations will play an important role in the implementation of industrial ecological ideas.

WHAT PLANNERS DO

Spatial planners deal with the substantive and procedural aspects of public policies affecting land use, buildings, transport systems and other infrastructure elements. Planners are also often involved in local economic development. Although the same university departments often teach economic planning and social policy courses, the traditional focus of the profession is on the built environment at the local level. Thus planners may design neighborhoods, specify permissible uses of land, lay out street patterns and transport corridors, locate community facilities such as schools and parks, establish aesthetic standards and ensure that private development proposals conform with a locality's grand vision, its comprehensive plan.

Historically, planning has roots in architecture, engineering, public health and late 19th-century progressive social movements (Campbell and Fainstein 1996; Duany *et al.* 2000). Architects interested in design problems on a larger scale than the single building began to focus on relationships among buildings. Engineers interested in the design of infrastructure elements and public utilities wanted to coordinate their projects with housing and land use initiatives. Public health advocates sought to reduce the impacts of pollution by separating noxious industries and residential neighborhoods. Social progressives hoped that, by providing urban workers with dignified, safe and pleasant housing, pathological behaviors might be averted. Planning became recognized as a distinct profession in 1914, with the founding of the British Royal Town Planning Institute, and recognition came later in most other countries (Masser and Yorisaki 1988). The field thus has both a technical and a normative basis.

In capitalist countries, planning sits at the intersection of market, political and administrative decision realms. Market actors finance a majority of the buildings, and thus take the risks and reap the rewards of real estate development. Government, meanwhile, finances a majority of the infrastructure elements and establishes regulations for land and building markets. Some decisions, such as the layout of the street grid and specification of allowed land uses, are usually under local control. Many other decisions, such as the placement of transport corridors and subsidy arrangements for housing finance, typically belong to higher levels of government. Thus intergovernmental relations and public–private sector relations are key elements of planning.

Table 38.1 shows illustrative data on selected physical planning characteristics for a few advanced industrial countries. The table confirms that circumstances differ radically even among countries at a similar level of economic development. Conditions diverge even more in the developing world. Illustratively, the World Bank (2000) reports that the urban percentage of total population in 1998 was above 75 in all of the countries shown in Table 38.1, but it was 11 in Nepal, 80 in Brazil, 31 in China, 28 in India and 42 in Nigeria. The total annual percentage population growth between 1980 and 1998 averaged well under 1.0 in most of the countries shown in the table, but it was 2.5 in Nepal, 1.7 in Brazil, 1.3 in China, 2.0 in India and 2.9 in Nigeria.

Cities also differ greatly from one another, even within the same country. According to the World Bank (2000), the percentage of work trips by public transport in 1993 was 67 in Rio de Janeiro, 53 in Delhi, 54 in Lagos, 27 in Copenhagen, 40 in Paris, 17 in Cologne, 22 in Amsterdam, 51 in New York and 20 in Atlanta. Residential crowding, measured in square meters of floor space per person in 1993, was 19 in Rio de Janeiro, 7 in Delhi, 6 in Lagos, 44 in Copenhagen, 30 in Paris, 34 in Cologne, 38 in Amsterdam and 41 in Toronto. The percentage of urban households with sewerage connections in 1993 was above 98 in Copenhagen, Paris, Cologne, Amsterdam, New York, Atlanta and Toronto, but only 87 in Rio de Janeiro, 40 in Delhi and 2 in Lagos.

Focusing primarily on advanced industrial economies, Downs (1999) identifies factors that influence metropolitan area development and lead to different outcomes in each country:

- size and cultural unity of the nation, which promote personal mobility and correspondingly reduce loyalty to places;
- national abundance of land, which affects its intensity of use;

Table 38.1 Comparisons of physical planning characteristics

		Denmark	France	Germany	Japan	Netherlands	Russia	UK	USA
Land area (1000km)^2[a]		42	546	350	375	34	16 996	242	9 159
Population (millions, 1998)[b]		5	59	82	126	16	147	59	270
Average annual population growth rate % 1980–98[b]		0.2	0.5	0.3	0.4	0.6	0.3	0.3	1.0
Urban % of total population[b]	1965[c]	77	67	79	67	86	52	87	72
	1980[b]	84	73	83	76	88	70	89	74
	1998[b]	85	75	87	79	89	77	89	77
Housing tenure circa 1980[d]	% owned	56	55	40	62	44	<30	61	65
	% private rent	21	30	42	23	14	0	12	30
	% social/public rent	23	15	18	15	42	>70	27	5
Cars per 1000 population, 1994[e]		312	430	492	341	383	46	410	727
Carbon dioxide emissions (kg/$GDP, 1996)[b]		0.5	0.3	0.5	0.4	0.5	1.5	0.5	0.7

Notes:
[a] US CIA (1999).
[b] World Bank (2000). Note that the definition of 'urban' varies slightly by country, so the urban percentage values are not strictly comparable.
[c] World Bank (1987). Germany value is for the Federal Republic only; Russia value is for the USSR.
[d] Maclennan (1999, p. 517), except for the Russia figures which are estimated for the USSR based on Struyk and Kolodeznikova (1999). In the post-Soviet Russia, much existing housing has been privatized but very little new housing has been built, heavy rent subsidy schemes remain in place and current statistics are not available (Struyk, 1996).
[e] US Department of Transportation (1997). Includes light trucks. This source does not report Netherlands data, so a comparable figure comes from European Union (2000).

- differences in governmental structures, which allow or preclude the formulation of national land use policies and metropolitan-scale governing;
- differences in political traditions, which support socialist or *laissez-faire* policies;
- nationally dominant cities, which encourage urbanization;
- impact of catastrophic (wartime) damage, which provides political impetus for new buildings and for planning;
- demographic drivers including natural population increase, needs of the oversized post-war baby boom age cohort, migration from rural to metropolitan areas, migration from abroad, migration out of central cities into suburbs, and interregional migrations;
- sociological factors including racism and the entry of females into the workforce;
- economic factors including the 1970s oil shocks and inflation, the sustained prosperity of the 1980s and 1990s, and the over-exuberant real estate markets of the late 1980s; and
- technological factors including differences in the creation of major highway networks, expanded use of automotive vehicles, expansion of air transport and airports, productivity increases in manufacturing and agriculture, plus telecommunications and computer innovations.

Just as physical characteristics differ from place to place, so planning practice varies. Table 38.2 summarizes key features of the context and institutional framework for planning in several advanced industrial countries for which there was access to data. The table's qualitative ratings, such 'high' and 'low', are based on my interpretation of the literature and may be idiosyncratic. Planning in developing nations is typically less formal, more opportunistic and more resource-constrained than is the case for the countries shown.

In some places, such as the USA, planning has become primarily a regulatory function: government planners working for localities enforce zoning regulations in a reactive manner. Consulting planners periodically help municipalities update these regulations while working on a more regular basis for developers to design projects and guide them through the regulatory process. In other places, such as Denmark or the UK, government planners tend to take a more active role in shaping urban form, by offering a grand vision and then negotiating incremental projects with developers in a relatively independent manner, although even here planners are tightly constrained by local politics (Flyvbjerg 1998; Healey *et al.* 1995). In Soviet Russia, where markets were non-existent, the supposedly powerful planners still had to negotiate for resources with other sectors and levels of government. Everywhere, planners balance substantive concerns about good design with procedural concerns about a legitimate planning process.

Here we briefly discuss the US case to illustrate the concepts embedded in Table 38.2. US planners place an emphasis on procedural legitimacy that often comes at the expense of good design – most US planners do little more than use their highly transparent regulatory process to weed out truly bad development proposals (Brooks 1988). This was not always the case. At the end of the 19th century, principles of town planning still widely accepted in Europe were also the US standard (Fishman 1996). For example, there was a broad consensus that neighborhoods should be designed to ensure that key destinations such as shopping, schools, religious buildings and transport connections were within a five-minute walk of most residences (Duany *et al.* 2000). The importance of separating

Table 38.2 Comparisons of planning contexts and institutional frameworks

	Denmark[a]	France[b]	Germany[c]	Japan[d]	Netherlands[e]	Russia[f]	UK[g]	USA[h]
Government structure	unitary	unitary	federal	unitary	unitary	federal	unitary	federal
National planning legislation	yes	yes	yes	yes	yes	yes	yes	no
Extent to which development is plan-led	high	high	high	low	high	formerly high	high	low
Local autonomy	medium	medium	high	medium	medium	formerly low	low	high
Local fiscal burden	low	medium	medium	high	low	formerly low	low	high
Government financial involvement in development	high	medium	medium	medium	high	formerly high	high	low
Extent of intersectoral and interregional coordination	high	high	medium	medium	high	formerly high	high	low
Administrative flexibility	medium	medium	medium	high	low	formerly low	high	high
Major themes	protect rural areas, participatory democracy	protect rural areas, design, litigation	decentralization, litigation, accountability	protect rural areas, spur private development	public property, consensus, detailed plans, certainty	difficult transition from central planning	development as a privilege not a right, green belts	localism, protect private property rights, separate land uses

Notes:

[a] Based on Edwards (1988); Summers *et al*. (1999).
[b] Based on Punter (1988); Summers *et al*. (1999).
[c] Based on Hooper (1988); Summers *et al*. (1999).
[d] Based on Masser (1988); Masser and Yorisaki (1988); Hirohara *et al*. (1988); Hebbert and Nakai (1988); Lo and Yeung (1995); Summers *et al*. (1999).
[e] Based on Davies (1988b); Needham and Verhage (1998); Summers *et al*. (1999).
[f] Based on Davidenko (1990); Davies (1990); Underhill (1990); Hartog (1999); Struyk and Kolodeznikova (1999); Summers *et al*. (1999).
[g] Based on Davies (1988a); Davies (1998); Summers *et al*. (1999).
[h] Based on Summers *et al*. (1999).

factories ('dark, satanic mills') from residential areas was also widely accepted. Citizens, civic leaders and developers agreed on such principles, and many at the beginning of the 20th century supported the 'City Beautiful' movement that laid a foundation for the new profession (Wilson 1989, pp. 281–305). However, once the US love affair with the car began, the consensus on planning principles fell apart. Following World War II, the federal government subsidized highway building and suburban home building on a massive scale, and many Americans lost their pedestrian orientation (Fishman 1999). Planners retained a consensus only on separation of uses, which became codified in zoning maps specifying single-use areas for shopping, industry, residences and recreation. Regulatory interpretation and enforcement in an adversarial setting became the planner's main job, and design for the various sub-markets was left largely to the developer.

Only recently have US planners become widely aware that single-use zoning codes make the mixed-use development patterns of their beloved traditional towns and walkable cities illegal. A neotraditional New Urbanist movement is now trying to rewrite the codes to encourage development that is more compact, allows mixed uses, follows traditional aesthetic guidelines and, above all, supports pedestrian movement (Leccese and McCormick 1999). Urban designers are rediscovering principles for integrating private properties into a non-commodifiable public whole; these include pleasing proportions and arrangements (form), ease of navigation (legibility), fine-grained multiple uses (vitality), purposeful thematization or identity creation (meaning) and elements that deliver intimacy and security (comfort) (Sternberg 2000).

Since authority for regulating land use in the USA has been delegated to the lowest possible level – the municipality – there is often poor regional coherence (Purcell 1997). Metropolitan-scale planning is a relative rarity in the USA except in the transport sector; hence land use–transport interactions are often driven by the road builders. Among the innovative exceptions are Portland, Oregon, which coordinates land use and infrastructure decisions within a regional growth boundary, and Minneapolis, Minnesota, which annexes new suburbs to keep regional growth within the same municipality. Alternatively, the state may step in: examples include New Jersey, which engages local, county and state planners in a cross-acceptance process to improve regional coordination, and Florida, which requires every locality to create a comprehensive plan and submit it for state-level approval.

Planning elsewhere in the world differs in many details from the US case, as shown in Table 38.2. But key features are consistent: negotiating between public and private interests, balancing between substantive and procedural concerns, working within an intergovernmental system, minimizing land use conflicts and improving the built environment.

PLANNING PRACTICE AND IE

The IE vision is being implemented at various levels, and planning practice affects several of them. Here we briefly discuss the key intersections at the macro, meso and micro levels.

Urban Spatial Patterns

At the macro level, prescriptive IE seeks to rationalize aggregate materials and energy flows, which are strongly influenced by settlement patterns. Alberti (1999) uses four dimensions to characterize these patterns in her review of links between urban patterns and environmental performance: (a) urban form (degree of centralization or decentralization); (b) density (ratio of population or jobs to area); (c) grain (diversity of functional land uses such as residential and industrial); and (d) connectivity (extent of interrelation and availability of multiple modes of circulation for people and goods among local destinations). Her review of the empirical literature suggests the following tentative relationships:

- resource use per capita or job decreases with centralization, higher density, finer grain and higher connectivity;
- air pollution concentrations have an indeterminate relationship with urban form, density and grain, and they decrease as connectivity improves;
- water pollution concentrations have an indeterminate relationship with urban form, grain and connectivity, and they increase as density increases;
- habitat fragmentation decreases with centralization and higher densities but it increases with better connectivity;
- exposure of human populations to air pollutants increases with density, and its relationships to urban form, grain and connectivity are unknown.

Planning is a major point of leverage on urban patterns, and hence is an important point of entry for industrial ecologists seeking to reduce aggregate environmental impacts. Local planners can be encouraged to support compact, mixed-use developments that allow residents to substitute walking for driving on some daily trips, for example (Leccese and McCormick 1999). Higher governmental levels need equal attention, of course – a consensus list of the top 10 influences on the US metropolis during the last 50 years awards its top spots to the popularity of cars, the 1956 federal Interstate Highway Act and Federal Housing Administration mortgage regulations (Fishman 1999).

Symbiosis in Eco-industrial Parks

At the meso level, eco-industrial parks (EIPs) fall squarely within the planner's domain. Inspired somewhat inappropriately by the Kalundorg symbiosis (see Chapter 27), a number of industrial park developers are describing their projects as EIPs. The range of initiatives includes full-scale symbioses, industrial parks with shared environmental management infrastructure, office parks with green design features such as walkability, industrial parks housing green firms that produce products such as solar cells, and even virtual ecoparks that merely facilitate materials exchanges among firms in a region (Côté and Cohen-Rosenthal 1998; Deppe *et al.* 2000). EIPs with co-located firms are more strongly affected by planning practices than those consisting of virtual networks of firms. A successful EIP-based economic development strategy will have much in common with traditional site development strategies; for example, projects will fare better when they have anchor tenants such as power plants (Chertow 1999b). Planners will typically focus on the following:

- ensuring that EIPs comply with local regulations;
- packaging EIPs as candidates for brownfield sites and as a component of local economic development efforts;

- assessing the net local benefits of EIPs, given that they require significant investments in transport and public utilities infrastructure;
- deciding to support EIPs actively, since their viability may depend on local fiscal incentives or even public ownership; and
- providing assistance to EIPs in surmounting regulatory barriers imposed by all levels of government.

Five major regulatory issues may arise (RTI 1996a). First, environmental regulations (for example, US PL 94–580) may classify secondary materials as hazardous wastes, making use and transport much more difficult and expensive. Next, it is common for regulators to count the individual enterprises within the EIP as pollution sources rather than allowing the EIP to optimize operations within a larger bubble. A tradition of single-medium permitting (that is, receiving official permission to emit a specified quantity of pollutants) serves as a further deterrent; regulators instead should adopt a multi-media approach to encourage firms to take a systemic view of their materials flows. Fourth, participants can be scared off by liability concerns (first, the liability associated with using secondary materials that are classified as hazardous wastes; and second, liability of separate companies when regulated under the same umbrella as other firms in the EIP). Finally, in the automotive era, land markets usually favor cheap and flexible greenfield sites even though brownfield sites are usually socially preferable. Therefore government may need to provide incentives for EIPs to use brownfields, such as waiving the purchasers' liability for earlier pollution.

Firms participating in EIPs – which are planned developments – must comply with both the government's external codes and the developer's internal covenants. External codes include zoning regulations, historic preservation standards, environmental regulations, building codes and other laws protecting public health and safety. Codes often have procedural requirements for public comment and agency review. Covenants, conditions and restrictions (CC&Rs) are often employed by developers who intend to subdivide and/or lease portions of a large tract of land and want to prohibit certain activities (for example, unauthorized uses) and require others (such as maintenance) by subsequent owners and tenants. EIPs typically require detailed CC&Rs, an appropriate governing body and an enforcement mechanism to ensure that they do not deteriorate into traditional industrial parks. For model codes and covenants, see CWEI (1999).

Although codes and covenants influence EIP development, the biggest issues are probably not regulatory. The greater challenges are those of finding complementary tenants, sparking local interest and financing an unfamiliar development concept (RTI 1996b). Entrepreneurial planners – developers – with a creative vision rooted in the concept of sustainable development will need to lead the charge, while the task of regulatory planners will be to judiciously blend facilitation and quality control.

Decomposers

At the meso level in IE, individual species become visible. Scavengers, recyclers, re-users and remanufacturers are important actors in many industrial ecosystems, and their ability to act depends to a large extent on governmental solid waste management policies. Planners influence solid waste management practices by virtue of their prominent roles in siting roads, landfills, incinerators, transfer stations, materials recovery facilities and other industrial land uses. Waste-processing and disposal sites often have LULU (locally undesirable land use) status, making them difficult to site, and thereby creating economies of scale. Single-use zoning laws may force scrap dealers and other small-time decomposers to locate far from their sources of scavengeable material, thus reducing their economic viability. In addition, the economic cost of waste collection and recycling is strongly influenced by the density and pattern of settlement. Nevertheless, planners play a relatively minor role relative to other governmental actors in solid waste management. More important are the municipal officials who supervise waste collection and recycling activities, and the higher levels of government that establish the rules of the game.

Eco-efficiency

Also at the meso level, planners affect eco-efficiency in important ways, although few know the concept or use it as a decision-making criterion. Current settlement patterns are inefficient in ways anticipated in the macro-level discussion above, but they bear listing here to clarify what planners can and cannot do. Eco-efficiency is a function of the following elements of the built environment:

- transport system design, including network layout, technology choices and modal split, all areas of interest to planners;
- water, sewer, solid waste and energy infrastructure elements, all of which are more cost-effective when settlement patterns have high density, and all of which are the concern of planners;
- ability to use district energy systems (which may include cogenerated electricity, heating and cooling, and which require compact settlements); these are rarely seen outside central cities in the USA, although they are more common in northern Europe;
- site designs and building orientations, which are a common focus of planners;
- building construction standards, which typically are not governed by planners but instead by building codes administered by a different department of local government; and
- appliance efficiency standards, which also are not governed by planners – they are typically set at the national level.

In short, planners have leverage over some but not all aspects of the built environment, and industrial ecologists should make appropriate efforts to introduce the eco-efficiency concept into the planning process, whether formal or informal. Local sustainability initiatives are one vehicle for this which has shown mixed success (Berke and Conroy 2000).

Biocompatibility

As concern over dissipative uses of toxic materials has grown (for example, Ayres 1989a), so too has interest in the biocompatibility of materials. Biocompatibility has two dimensions: first, that the material is safe for humans and ecosystems in its intended use; and second, that upon disposal it biodegrades or otherwise becomes harmless. Biocompatible materials are being introduced slowly into the built environment, primarily through the efforts of innovative architects and engineers working on green buildings (Calmenson 1997). However, planners can encourage this trend, especially in the domains of site and transport planning, where innovative paving materials are becoming available.

Behavioral Incentives

At the micro level, 'better' behavior can reduce the need for LULUs such as highways, power plants and landfills. The behavior of individuals has not been a focus of IE to date, but it should become so because of the opportunities behavior changes present for environmental improvement (Fischoff and Small 1999). In contrast, as social scientists have penetrated the ranks of planning departments at universities (Goldstein 1997), planners have gained expertise in analyzing the appropriate uses of behavioral incentives and in designing appropriate policy interventions. Common applications include the following planning and public policy actions:

- privatization, deregulation and outsourcing of public services to bring these high-impact activities under market discipline (Stein 1990; Roth 1987);
- imposing user fees on trash disposal, water and sewer use to send economic signals upstream to households, rather than continuing to pay for these services from general tax revenues (Miranda *et al.* 1994);
- imposing impact fees on developers to cover the negative fiscal impacts (due to increased demand for public services) of new buildings (Brueckner 1996);
- adopting green procurement criteria when purchasing government supplies (Lyons 1999), and awarding economic development grants to green start-ups (NJCEGC 2000).

Behavioral incentives need not be economic, although that is the focus of the literature. Practical exhortations to recycle or buy green products often simply employ moral arguments.

Applied Utopian Endeavors

Much like IE, the substantive side of planning has comprehensive, often utopian technical aspirations, manifested most clearly in the creation of master plans that lay out transport corridors and specify locally allowed land uses. Substantive planners have also attempted many activities in past decades that are now being reinvented by industrial ecologists, including city systems analysis (Batty 1994) and the pursuit of 'optimal' urban designs (Fishman 1977).

Yet comprehensive planning lost some of its credibility following a series of well-intentioned yet ill-conceived and brutal interventions in the urban landscape. Le Corbusier's 'tower in a park' concept, for example, was embraced for its cost-effectiveness by both Soviet and US planners of public housing projects. While it led to highly cost-effective – but ugly – housing for many urban Russians, the results were disastrous in the land-rich eastern areas (Underhill 1990). In the USA, this approach had the effect of warehousing the poor in huge, frightening, isolated concrete boxes after destroying their old, well-connected, human-scale neighborhoods (Jacobs 1961). Also infamous were arrogant transport planners such as Robert Moses of New York, who rammed highways through urban neighborhoods, displacing thousands of residents without solving traffic problems, and starving other transport modes in the process (Caro 1974). Even the systems analysts lost credibility for making models that were overly ambitious and unrealistic; critics coined words like 'hypercomprehensive' to describe their failings (Lee 1973).

As planners in North America and western Europe left behind the hubris of the 1950s and 1960s, intellectual effort in planning shifted to procedural topics, with some arguing for an advocacy stance, others for a conflict resolution stance, and the current consensus shifting towards a collaborative stance (Guhathakurta 1999). Legitimacy and professional incrementalism became the watchwords. Only recently, with the growing concern over global megacities and sprawl development, has boldness of a modest sort again become seen as a virtue (Myers and Kitsuse 2000). In short, today's planners strive to balance the substantive and procedural aspects of their work.

Industrial ecologists would do well to recognize that their field too includes both substantive and procedural challenges (Andrews 1999b). Like planning, prescriptive IE is in danger of cycling from hubris to despair before discovering humility and effectiveness. Debates over earth systems engineering, for example, represent healthy efforts to address this challenge in advance (see Chapter 46). Until the grand visions of IE are tempered by implementation experience, they will remain utopian, and will be a poor basis for public decisions. Application is a great if humbling teacher, and we should reflectively practice IE in order to learn from it.

39. Industrial estates as model ecosystems

Fritz Balkau

Natural ecosystems have evolved over many years to have an apparent stability and organized functioning that we would like to see in other areas of our lives. There has been a frequent desire by mankind to learn from, and even mimic, natural systems, especially as our current economic growth models are starting to show some limitations. The study of industrial ecology can be seen as an attempt to view our modern production methods in such a 'systems' context, where the individual actions of various producers and consumers are played out in a larger framework of mutual interdependence.

Natural systems have no objective other than the survival of the individuals within them, and it can take many generations to achieve a stable state. In an industrial system the ecological concepts of symbiosis, interdependence and competition have to be seen against the relatively short time scale of the human management objectives of the same system. Regular changes arising from external policy shifts, technology development, regulatory change and a rapidly evolving external economic 'climate' can change dramatically within the lifetime of an individual enterprise. The optimization of such a system in a reasonable time scale can hardly be achieved by a natural evolutionary process; rather it must rely on guidance from management decisions by an authoritative body. But the central planning and management of all environmental initiatives and dependencies is beyond the capacity of any institution. The challenge for a managed industrial ecology is one of arranging for an improved framework within which rational environmental decisions can take place, leaving room for natural selection at the micro level to forge the necessary day-to-day relationships.

Among the more stable entities in our current industrial landscape that could create the necessary framework for an evolution of synergies are industrial estates and export-processing zones, which by their very nature have some expectation of longevity as managed development areas. This chapter looks at the potential of industrial estates as a place where industrial ecology concepts could be rationally implemented and promoted. Many estates have enough enterprises to allow for a potentially large number of synergies or cooperative actions to be easily arranged. Because of their pivotal role in planning and operation of an industrial area, estate managers are in principle able to act in an interventionist fashion to encourage or even require certain forms of collective environmental behavior. But, as most existing estates are commercial entities in their own right, the challenge is also to make industrial ecology an attractive business proposition (or a survival factor) for them. One way of achieving this is to stress the notion of eco-efficiency on a large scale, and the attractiveness of achieving environmental compliance at lower cost by deliberate intervention to foster cooperative behavior.

Growing national awareness in all countries of the need to better protect people and our resource base means that estates, as indeed industrial development at all levels, are strongly influenced by national environmental regulations. But environmental policy is now increasingly developed in the context of a broader 'sustainable development' idea, which tries to integrate environment protection also with economics and social advancement in a more balanced way. To better understand the environmental policy context of industrial ecosystems, we need first to review the influence and impact of the sustainable development agenda on industrial production overall.

SUSTAINABLE DEVELOPMENT OBJECTIVES IN INDUSTRIAL ECOSYSTEMS

The framework for sustainable development was developed at the Earth Summit in Rio de Janeiro in 1992 through a document entitled 'Agenda 21' (United Nations 1992). Since this 'blueprint for development' as it is sometimes called was prepared, a number of the key issues have further evolved, for example those of climate change and biodiversity. There has been a shift in emphasis in environmental programs to address the *causal factors* of environmental degradation such as poverty, inefficient industrial production processes and excessive resource use. Also more attention is now placed on the *demand side* of our production systems, by focusing on the design of products (from an environmental point of view) and on the globally unsustainable consumption patterns of the wealthier parts of our society.

While Agenda 21 and even the notion of sustainable development may seem abstract and unimportant to some, it has in fact exerted considerable influence on the way national development plans are prepared in many countries. Development banks and multilateral agencies have restructured their criteria around the priorities described in Agenda 21. Even business groups such as the World Business Council for Sustainable Development have used the terminology and issues in their programs and outreach.

From the point of view of managing industrial ecosystems, the key issues that emerge from Agenda 21 are:

- reduction in the amount, and safe management, of wastes and emissions from industrial production and users of products;
- protection of natural habitats and species diversity, including living resources needed for food production;
- protection of oceans and seas, including especially coastal zones;
- release of greenhouse gases (GHG) and ozone-depleting substances (ODS);
- availability and conservation of soil, water and energy;
- appropriate choice, safe production, management and use of chemicals;
- landscape and heritage site protection and human amenity.

One immediate implication for industrial ecology thinking is that the environmental agenda is now no longer dominated by issues of materials and energy flows. Biodiversity, habitat and heritage protection, global issues of climate change and the management of small quantities of ecologically significant contaminants have assumed a greater prominence in environmental policy everywhere. These new priorities have to be taken into account in our management of industrial ecosystems, along with the more familiar ones of pollution and waste. More emphasis is also being given to addressing the fundamental causes of pollution (sometimes called 'driving forces'), and in promoting new management and regulatory instruments rather than simply addressing waste flows through recycling. These newer issues are already reflected in the current priorities of environmental agencies, development assistance programs and international law (which eventually becomes translated into national law), and are reflected in updated thinking about the environmental impact assessment (EIA) process on which many of these mechanisms depend.

Agenda 21 already described some of the *mechanisms* that would be needed for implementation. New ideas such as cleaner production, dematerialization, life cycle product management and sustainable consumption are now starting to influence both government and business policy. What is still needed is a clearer conceptual framework for their application. A particularly useful and succinct description of this was given by Erkman (Erkman 1998). An important point made by Erkman is that continuous improvement is not enough. But many of the wider 'system changes' needed for the future are difficult to achieve without a high degree of cooperation among the social and business partners. This means that mechanisms for collective management are at least as important as those at the level of individual enterprises. Concepts such as environmental supply-chain management are thus vital to the reorientation needed by the industrial ecosystem. The recent growth in voluntary codes, covenants and agreements (UNEP 1998) is evidence that this need for cooperative action has been well appreciated by at least some business sectors of our society, although it still needs considerable further extension. The importance of collective measures is especially important in the context of management options for industrial estates.

The breadth of the sustainable development concept makes it a useful framework for a comprehensive 'ecological' approach to industrial development. In particular, it allows the incorporation of a number of previously disparate environmental issues into the core considerations of industrial ecology. Many of the components of an 'ecology' approach to industrial production have been extensively described in published articles and conferences, as referenced elsewhere in this handbook, but there has been less detail on how these components could be assembled, by whom they could be implemented and what mechanism could be used to bring them to fruition. Accordingly, studies of industrial ecology – and the dynamics that it implies – need to be supplemented by studies of the way industrial systems can be administered.

In a comprehensive ecological approach it is important to ensure that *all* the sustainability objectives are addressed simultaneously. Limiting the focus on one or two, such as waste or energy, is only a partial solution to the environmental challenge, and is in any case unlikely to meet the wider regulatory and public demands now faced by industry. For example, the mining industry is subject to enormous pressure to address issues of biodiversity, site remediation, metals toxicity in products, employment and human rights. The way these issues are resolved will strongly influence the options for waste and energy management, and the optimum solution will often be a compromise of objectives.

Industrial estates come in a great variety of types and sizes. Surveys (UNEP 1997) have estimated the number of estates worldwide at well over 20000. In China in 1998, over 10 per cent of foreign investment occurred in export-processing zones (UNEP 1999). Very large estates, such as in Suzhou, China, or the Jebel Ali Free Zone in Dubai, are effectively new industrial cities with up to half a million people, covering over of 50 square kilometers of land. The smallest are simple activity zones on the outskirts of a provincial town, managed by the municipal administration. Elsewhere we can mention the old heavy industry complexes in many countries, degraded industrial zones striving for renewal (now sometimes called ‘brownfield’ sites) and smaller groups of companies establishing themselves independently in economically attractive locations.

All the usual environmental problems concerning industrial production manifest themselves in an estate: wastes, emissions, energy use and chemical contamination, to name only the more prominent ones. But the size of industrial estates often also brings into play a range of additional environmental issues not immediately relevant to individual companies, such as habitat and biodiversity impact, water catchment and airshed management, landscape and land-use issues, management of natural resources such as groundwater, and location of waste disposal facilities. For example, Burnside Park in Canada occupies a significant catchment area in a wet climate where erosion, run-off and natural habitat have to be managed to avoid impacts on downstream communities.

These ‘new’ issues require the creation of environmental infrastructure functions such as erosion control, catchment management, waste collection, fire and emergency services, traffic management, energy supply and bulk chemicals storage. These are not usually part of the corporate agenda of companies; rather they are the domain of collective entities such as cities or regional governments, and their application often requires policy guidance and organizational management rather than numerical standards. Yet they are a new but essential part of an industrial ecology approach (*Décision Environnement* 1994; RIET 1996).

In applying such solutions, estates have a natural advantage over private corporations. Because of their size, and especially when they are in public ownership, many estates remain more tightly bound to government policy on environment and development than smaller production units. In Dubai, the Jebel Ali Free Zone was able to negotiate the joint development of municipal hazardous waste facilities to allow it to adequately solve its own problems of toxic industrial waste (personal communication). An independent resolution of its own situation (by, for example, waste reduction or recycling) would have affected the viability and size of the municipal facilities being planned. Policy dependence is not limited to environment. The size of many estates places an extra burden on local infrastructure such as water supply, transport capacity and emergency services, and requires coordination of estate management with more regional development plans.

This, and the fact that their timetable of investment return is longer than that of many of their tenants, means that estates are better able to address the long-term aspects of a sustainable development agenda. The combination of policy dependence and longer-term vision, when combined with an understanding of the opportunities of cooperative management, gives them the possibility of some creative solutions in industrial ecology that have not so far been tried on a large scale. As we will see below, these options concern both the planning phase when infrastructure is being created and the subsequent operations stage when the estate starts to influence the environmental behavior of individual enterprises.

However, there are also some factors holding back such management initiatives. Industrial estates are often in a competitive situation among themselves, and are looking for (or at least not to lose) commercial advantage. Lack of environmental vision among land developers means that social and environmental factors are still often regarded as constraints rather than opportunities. There is little evidence that the majority of estate planners at present see much advantage in collective environmental action beyond providing tenant companies with waste disposal services to allow compliance with national pollution standards. This is despite the fact that cooperative industrial action is one of the remaining untapped sources of further improvement in manufacturing efficiency for enterprises, after internal improvements and process optimization have been completed.

MANAGEMENT OPTIONS FOR INDUSTRIAL ECOLOGY

To accelerate the adoption of ecosystem approaches, it is useful to study how, within a competitive context, an estate owner could create practical opportunities and incentives for companies to act in concert. A number of examples are found in the UNEP report (1997); however, the scientific literature has so far focused more on studying synergies at the plant level rather than at the macro level of an entire estate. Nevertheless, several case studies of pioneering efforts are given at the end of this chapter. Overall, there is a need for a better sharing of experiences and synthesis by creating a network of estate managers who can work together to identify the most fruitful options. In recognition of this, UNEP has held several workshops for estate managers in Asia (UNEP 1997) and is now working with the Chinese government to plan a number of pilot studies in estates in China (UNEP 1999).

The potential action by estates for collective management is at two levels. The first is the planning of the estate to allow better environmental performance of the complex as a whole. This would aim at a physical layout that is in harmony with the terrain, planning of transport and chemical storage infrastructure, provision of energy-efficient buildings, provision of appropriate services and energy (including renewable energy), planning for juxtaposition of factories to improve sharing of surplus energy and wastes, establishment of a recycling center, reserving vacant land for natural habitat, and so on.

The second possibility is promoting organizational procedures to facilitate better environmental management at the plant level, establishing mechanisms for sharing of services and surplus wastes, and creating information and communications systems for companies to allow them to find their own individual solutions to environmental issues. These are facilitating roles to accelerate the evolution of greater synergies between individual plants, and are based on cooperation rather than coercion. Without such facilitating mechanisms, synergies and symbiotic arrangements can be slow to start.

The twin approach of planning and operation can assist in improving performance on a wide range of sustainable development elements. For example, waste disposal through common treatment facilities allows easier compliance in principle with national pollution standards. But most waste treatment plants are actually not well able to deal with the complex mixture of effluents from a large estate (most really only treat the organic portion). The proper dimensioning of such plants to work efficiently during the extended growth phase of an estate is also difficult. It is not surprising that many estates have difficulty in meeting national effluent standards in a cost-effective way. A more rational design and management approach could improve the performance of many such plants.

Solid waste is another management element. It is clear that in most estates the waste is currently moved off the site This arrangement merely allows the polluter to shift the cost burden somewhere else. A more significant 'ecological' achievement would be to move towards avoiding waste altogether through favoring tenant companies that adopt cleaner production technologies, higher rates of recycling and waste exchange, by supporting the production of waste-derived products such as energy, compost and building materials for sale in external markets. At the same time the estate needs to ensure that its own operations are optimized so as to produce as little waste and pollution as possible.

Environmental elements other than processing waste need also to be optimized. An effective energy management scheme for an estate (including recovery of waste energy) reaches beyond what individual companies can achieve. Heavy industries can be grouped into a 'utility island' in the center of the estate to facilitate transfer of waste heat to other clients. Energy generation plants can be designed from the outset for waste heat re-use elsewhere in the estate. Through a cooperative management of all greenhouse gases, including ozone-depleting substances, an estate could make significant contributions to national targets under the Kyoto and Montreal protocols. An integrated water management scheme can reduce dependence on external sources, with concomitant economic benefits to the estate and the region at large.

A land planning approach based on 'design with nature' principles can make a major contribution to conservation of biodiversity values through a more rational layout of the entire estate, and the deliberate creation of new wildlife habitats in buffer zones and unoccupied land (few estates occupy 100 per cent of land with structures). Such habitats can be managed by the estate, or through third party agencies and organizations.

In principle at least, an environmental action plan that incorporates the above issues would constitute a blueprint for a managed industrial ecology. It would identify the issues to be addressed, define the interactions and relationships between key stakeholders and describe the practical management options for the estate manager and outside authorities. In its most useful form, such a plan also becomes a useful instrument in guiding the environmental initiatives of individual enterprises.

The elements of such an action plan are described in the 1997 UNEP publication, 'Environmental Management of Industrial Estates'. Several high-level workshops with managers in Southeast Asia during 1997–2001 have been useful in promoting further understanding of the way corporate environmental management systems and tools can be applied by industrial estates.

To implement an environmental action program in a rational way, some estates have started to use environmental management systems such as ISO 14000 that have come into more common use elsewhere in the corporate sector (ISO 1995; Harrison 1995). Because the environmental agenda of an estate is necessarily broader than that of a company, such a management system should really incorporate the new elements mentioned earlier, such as landscape, habitat protection, water resource management and air quality. However, owing to unfamiliarity with environmental systems, many estates are still dealing with only a selected number of issues, for only a small portion of their estate (usually the operations they themselves manage). The unaddressed issues leave them open to the resurgence of compliance problems and difficulties with neighboring communities, as has already happened in Thailand for example, where controversial incidents of air pollution from a chemical factory were outside the responsibility of the estate managers (personal communication).

Most management systems so far developed for estates have omitted options for collective forms of environmental management in their action plans. This is a major deficiency, since collective response to issues of energy and water, chemical safety and emergency preparedness and response, waste re-use and recycling is far more effective than leaving this to individual companies. Environmental services, such as waste exchange, environmental training, information and news, have also been slow to appear. As a consequence, many organizational possibilities of rationalizing materials and energy flows have been overlooked, as have opportunities for addressing other sustainable development issues through some form of cooperative action.

Among the barriers to progress is the perpetuation of old-fashioned managerial attitudes to environment, and a lack of understanding of what the sustainable development agenda actually means for future business. Estate management and marketing is sometimes based on what the manager thinks prospective tenants believe about environment; that is, that it is a significant cost item. This misconception can go so far as to prevent estates from advertising their ISO 14001 certification because they feel that prospective tenants might believe that such a system must surely give the estate a high environmental cost structure. As a result of such perceptions, estates generally do not propose collective environmental solutions to their tenants for fear of discouraging investment. Few set out to promote environmental quality as a marketing device. One of the barriers to industrial ecology is indeed still the common belief that it is a burden rather than a business advantage.

EXAMPLES OF ENVIRONMENTAL INITIATIVES IN INDUSTRIAL ESTATES

The above discussion has so far been somewhat abstract. Have any of the principles been applied, and what have been the results? The best-known example of industrial symbiosis is that of Kalundborg (UNEP 1996, p.6; see also Chapter 27), in Denmark, although it is not a managed estate. Here a high level of synergy has been achieved by re-use of the waste flows between a number of industries. This situation was never 'designed in'; it has slowly evolved on its own over the years. In a new estate project the possibility of such synergies could be built in from the start. It should be noted that Kalundborg is rarely cited for maximizing other environmental outcomes such as biodiversity, hazardous chemical control, industrial safety or even product policy. The composition of other parts of the waste stream not currently recovered is unknown, a deficiency from a total waste management point of view. Despite its obvious success in achieving symbiosis in some areas of waste management, even Kalundborg has some way to go before it becomes a model 'ecology' from a sustainable development point of view.

Some estates have focused on managing a different ecology dimension. Rather than focusing on materials and energy, the ValuePark operated by Dow Chemicals in Germany (personal communication) has concentrated on cooperation in other business values such as buildings, transport, storage and sales, effectively creating synergies at the product end of the industrial cycle rather than at the waste end. In terms of industrial ecology, this may well be a sign for the future. New industrial plants are generally much cleaner than their older homologues, and offer fewer opportunities for synergies based on exchange of surplus materials.

An example of a comprehensive, traditional 'command and control' approach to environment in established estates can be seen in the Jebel Ali Free Zone Authority in Dubai. Here strict entry standards apply to industries wishing to locate there, and require an environmental impact assessment from each applicant. Permission to locate in the estate is contingent on the applicant demonstrating appropriate control and treatment of industrial emissions. Discharges and emissions are tightly regulated, and new waste disposal facilities – both for dry waste and for chemicals – have been provided under an arrangement with the Dubai municipality. As in most such estates, 'environment' is associated with the pollution agenda. The broader sustainability issues in Agenda 21 are not yet included in the estate's action plan.

The application of ISO 14000 by estates is a recent but rapidly growing trend. This can be seen especially in Southeast Asia, for example Batamindo in Indonesia, but also gradually in Europe, as with the Plaine de l'Ain in France. The Industrial Estate Authority of Thailand has the objective of eventually certifying all of the 20-odd estates under its control. Several private estates in Thailand have already taken steps towards certification. The environmental action plans of such estates cover for the moment only the services actually provided by the estate, such as common waste treatment and disposal and maintenance of the site. They do not include the operations of tenant companies where the majority of the pollution and waste is actually produced. With increased experience in the application of such systems it can be expected that, eventually, such estate management systems will also include other environmental issues, and perhaps even cover the environmental interaction between the estate and its corporate tenants.

A major environmental challenge for the estate remains the administrative and management mechanisms for achieving synergies across all operations (including tenants). The French PALME model (Orée 1995; UNEP 1997) has tried to address this through the prior (that is, before development begins) negotiation of a social contract with all relevant stakeholders. These stakeholders not only identify their own environmental objectives and concerns, but commit themselves to specific contributions to make the final plan work. The stakeholders include service agencies such as energy and transport, but also local civic groups and local government who accept the final negotiated compromise. The final written agreement includes an action plan listing the commitment of all stakeholders. Two recent estates in France, in Boulogne-sur-Mer and in Chalon-sur-Saône, are currently trying this approach of consensus building around environmental and social objectives.

As information remains the key to taking environmental action, the example of the Environmental Information Center in Burnside Park, Nova Scotia, Canada also merits attention (Côté 1994; Burnside at *http://www.dal.ca/eco-burnside*). The center is an independent initiative of Dalhousie University, supported by local government and the estate management itself (which here is part of local government). The center is staffed by several professionals who prepare information on waste reduction, recycling and management options, as well as other aspects of the environmental agenda. The staff works with individual companies as well as with the estate personnel.

The examples closest to policy-driven estate development are found in the 'brownfields' developments on degraded industrial sites in several North American locations (Cohen-Rosenthal, in UNEP 1996, p.14). The political imperatives of site rehabilitation and job creation in disadvantaged communities leads naturally to the concurrent adoption of a comprehensive environmental agenda for the estate development. A wide variety of policy outcomes have been formulated for such projects, including recycling, energy conservation, sustainable and socially useful products, re-establishment of natural areas and community health. The achievement of such a broad agenda is facilitated by the injection of public funds, although a range of management models has been found.

The above provides a number of different models in the direction of an industrial ecology approach through application of concrete management systems. No single estate has yet combined all these elements in a comprehensive way, with the result that the progress is still quite uneven.

In nearly all cases so far, the estate management has focused on actions over which the estate manager himself has full management control rather than actions by the companies in the estate. With the exception of the PALME model, there has been relatively little effort to explore the options for cooperative management. It remains to be seen how and when the estate manager elsewhere takes the initiative to provide environmental brokerage services to enable companies to achieve such synergies more quickly and more easily than is the case at present.

CONCLUSION

Industrial ecology is a useful framework for helping to achieve a more sustainable industrial development. As the sustainability agenda now includes a wider range of issues than previously, the current preoccupation of industrial ecology with materials and energy flows needs to be enlarged to encompass also issues such as global resource use, biodiversity and chemical safety.

Industrial estates have the potential to be important players in implementing the industrial ecology concept. Large estates contain a wide diversity of companies and service functions that allow the creation of considerable synergies if this were to be one of the objectives of an estate. More so than individual enterprises, they already have the administrative structures and managerial resources to be able to translate the industrial ecology concepts into actual practice by planning their estates to facilitate the creation of synergies among their tenants, establishing cooperative arrangements for environmental information and services, and providing key functions such as materials exchanges and waste minimization advice on an estate-wide basis.

The growing diversity and maturity of environmental policy instruments and environmental management systems means that the management tools are now available to do this. However, some of these tools need to be further adapted for use by estate managers, to take account of the specifics of their situation.

Up to now, industrial estates have not seen themselves as frontline advocates in this field, and their environmental actions have mostly focused on achieving compliance with pollution standards rather than contributing to sustainable development policy. There is a need to provide estate managers with information, incentives and policy direction to encourage stronger adoption of industrial ecology concepts. The industrial ecology community could usefully reach out to this hitherto neglected target group to provide the necessary awareness, support and training to enable a more proactive stance on the part of industrial estates.

40. Closed-loop supply chains

V. Daniel R. Guide, Jr and Luk N. van Wassenhove

There are numerous examples and cases available of products that are being re-used via remanufacturing or recycling, or combinations of re-use activities (Thierry *et al.* 1995; Krikke *et al.* 1999; Guide 2000; Toktay *et al.* 2000; Fleischmann 2000). However, these products and their supply chains are not all the same with respect to a number of critical dimensions, including product acquisition, reverse logistics, inspection, testing and disposition, remanufacturing, and distribution and selling of the remanufactured product. In the following sections we document a number of diverse products that are currently being remanufactured and describe their supply chains. After each case, we summarize and discuss the distinguishing features of the supply chains. Finally, we discuss the management of each of the different supply chain systems and identify the key research issues.

SUPPLY CHAINS FOR REFILLABLE CONTAINERS

Xerox Copy/Print Cartridge Return Program

Xerox introduced its program for copy/print cartridge returns in 1991 and, currently it covers 80 per cent of the toner/print cartridge line. In 1998, Xerox expanded the program to include the recycling of waste toner from high-speed copier and commercial production publishing systems. The return rate for cartridges in Europe and North America was greater than 60 per cent for 1998. This equates to over 2.86 million kilograms of material remanufactured or recycled just from cartridges. Xerox reports avoiding almost 23 million kilograms of landfill because of its re-use programs. The cartridges are designed for remanufacturing and recycling of materials not fit for remanufacture.

Customers return the cartridges by placing the spent cartridge in the packaging used for a full cartridge and attaching a pre-paid postage label provided by Xerox. The returned cartridges are cleaned and inspected, and then parts are re-used or materials recycled. The full cartridges are then distributed through normal distribution channels to customers. The final cartridge product containing remanufactured parts or recycled materials is indistinguishable from cartridges containing exclusively virgin materials. Figure 40.1 shows the supply chain, in simplified form, for a cartridge. Xerox is currently testing the use of 'ecoboxes' to allow bulk returns from high-volume users in Europe. Xerox will arrange for regular pick-ups of the boxes by its own carrier network. A bulk returns process allows each high-volume user to batch cartridges and may lower the returns costs absorbed by Xerox. (The information in this section was obtained from Xerox 1999.)

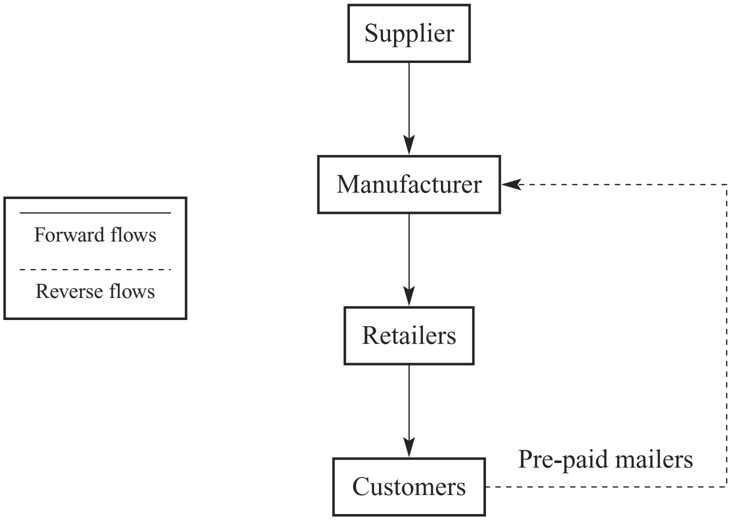


Figure 40.1 A closed-loop supply chain for cartridge re-use

Kodak Single-use Cameras

Kodak started its program to re-use its single-use camera line in 1990. The first stage of the program was to redesign the cameras so that parts could be re-used and film reloaded. The entire line of single-use cameras may be remanufactured or recycled and the amount of materials per camera that are re-usable range from 77 to 80 per cent. The second stage involved forging agreements with photofinishers to return the cameras to Kodak after consumers had turned them in for processing. Kodak now enjoys a return rate greater than 70 per cent in the USA and almost 60 per cent worldwide. Since 1990, Kodak has re-used over 310 million cameras, and has active programs in over 20 countries.

The process flow for the re-use of cameras, after the sale of the camera, starts with the consumer returning the camera to a photofinisher to develop the film. The photofinisher then batches the cameras into specially designed shipping containers and sends them to one of three collections centers. Kodak has entered into agreements with other manufacturers (Fuji, Konica, and others) of single-use cameras that allow for the use of common collection centers. At the collection center, the cameras are sorted according to manufacturer and then by camera model. After the sorting operations the cameras are shipped to a subcontractor facility where the cameras are stripped of packaging materials, disassembled and cleaned. Some parts are routinely re-used, some are removed (batteries are always removed) and the frame and flash circuit board are carefully tested. These sub-assemblies are then shipped to one of three Kodak facilities that manufacture single-use cameras. At the Kodak facility, the cameras are loaded with film and a fresh battery (flash models only) and finally fitted with new outer packaging. The final product is now distributed to retailers for resale. The final product containing remanufactured parts and recycled materials is indistinguishable to consumers from single-use cameras containing no re-used parts. Figure 40.2 shows the supply chain network for re-usable cameras. (The information in this section was developed from Kodak 1999.)

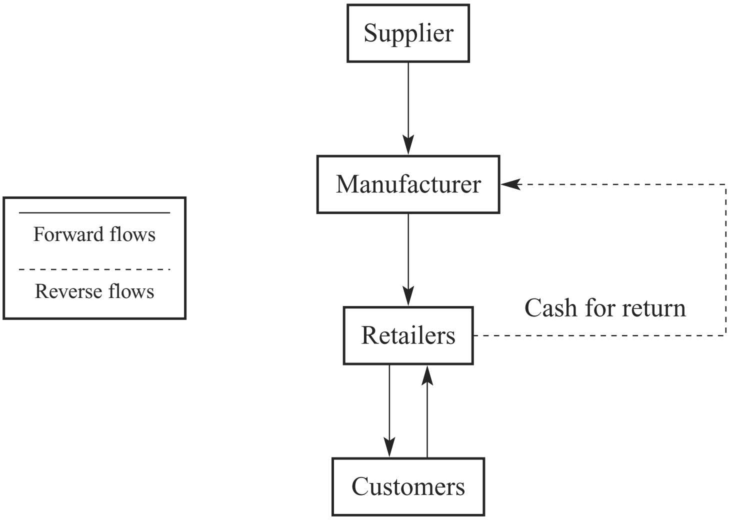


Figure 40.2 A closed-loop supply chain for single-use cameras

CHARACTERISTICS OF REFILLABLE CONTAINER CLOSED-LOOP SUPPLY CHAINS

These products, toner/printer cartridges and single-use cameras, are both containers required to sell the contents. Consumers do not distinguish between a new and a remanufactured/re-used product since the purpose of buying the container is to gain access to the contents, which are toner ink and film, respectively. Additionally, the customer also has no way of determining, aside from labeling, whether the container has been remanufactured or that it contains re-used materials. The returned containers in both cases are blended with new materials so there is no distinguishing between new and re-used products. This is possible because the technology is extremely stable for toner/print cartridges and single-use cameras. The markets for the remanufactured products are exactly the same as for the new products since the two are indistinguishable. The characteristics of the supply chain for container re-use are listed in Table 40.1. The volumes are very high,

Table 40.1 Characteristics of closed-loop supply chains for refillable containers

Commodity goods
Containers for consumables
High volume
Low variance
Non-distinguishable products
Simple products
OEM-controlled
Short lead times

in the millions for both products, and the annual returns quantities are stable or have a known growth rate. In both cases the original equipment manufacturer (OEM) controls the reverse logistics network and the re-use alternatives.

SUPPLY CHAINS FOR INDUSTRIAL REMANUFACTURING: COPYMAGIC

Industrial remanufacturing is mechanical item remanufacturing and accounts for the majority of remanufacturing operations in the USA (Guide 2000). Xerox reports that over 90 per cent of its copier equipment is remanufacturable and that using remanufactured parts has enabled Xerox to reduce landfill materials by over 300 million kilograms since 1998 (Xerox 1999). Xerox has redesigned its photocopier lines using modular design principles to allow for part and component re-use.

Thierry *et al.* (1995) describes in detail the product recovery processes used by a multinational photocopier manufacturer (nicknamed 'CopyMagic') and we present an overview of the re-use processes and the product characteristics. CopyMagic brings new products to markets and takes back used products from customers. The majority of the return flows are from expired lease agreements, although some used products may be purchased to ensure sufficient quantities of used products. After products are returned there are several alternatives for product re-use: repair, cannibalization, recycling and remanufacturing. A product may be used for any or all of the above re-use options. For example, a product may be disassembled and some parts cannibalized for use as spares, other parts may be repaired and re-used, some other parts are fully remanufactured, and parts worn beyond higher-order re-use are recycled. The most appropriate level of re-use for products, components and parts should be driven by economics: for example, the most revenue-effective re-use option.

CopyMagic does all remanufacturing operations in-house and uses a common production line for both new and remanufactured products. The use of existing production facilities greatly reduced start-up costs for remanufacturing, but has the disadvantage of complex production planning and control. Remanufactured products may be offered as technical upgrades, or with original technology, and this has enabled CopyMagic to exploit additional market segments. Product design relies heavily on modular design concepts to allow for interchangeable parts and components between models and product lines, in addition to allowing technical upgrades. Products designed for remanufacturing are also easier to repair and service, allowing for better customer relations. The firm has been able to reduce the number of suppliers because of common design platforms and components. A disadvantage to remanufacturing is that suppliers with better-quality parts may have lower sales volumes since their parts may be re-used more frequently.

The marketing of remanufactured products is complex and CopyMagic has had to tightly control the quality of remanufactured products and convince customers that the quality of remanufactured products is the same as that of a new product. However, the company has an enhanced image thanks to the 'green image' of re-used products.

The additional complexities of product re-use have caused CopyMagic to design and implement new information systems to forecast and track product returns, analyze the returns for yields and the condition of products, modules and parts, and to track the per-

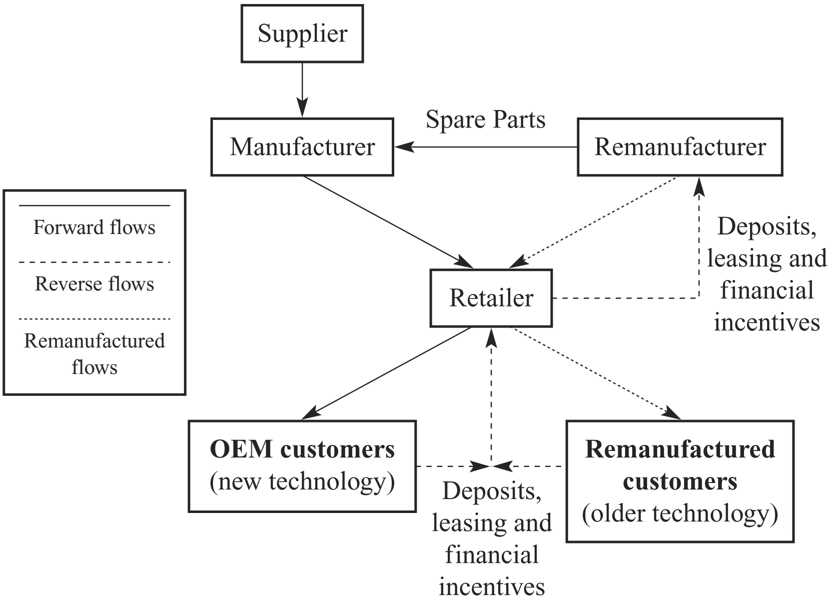


Figure 40.3 A closed-loop supply chain for photocopiers

formance of remanufactured units. The supply chain (illustrated in Figure 40.3) required for these activities is more complex to plan, monitor and control. In the figure, we show the remanufacturer as being physically separate from the manufacturer. In the case of third party remanufacturing, this is standard practice. However, Xerox and Ocè both use a common facility for new and remanufactured products. Where a common facility is used, new and remanufactured parts are mixed to produce end items. In this case the distinction between manufacturing and remanufacturing is not physical, but is used simply to illustrate the different flows of components and parts.

CHARACTERISTICS OF CLOSED-LOOP SUPPLY CHAINS FOR INDUSTRIAL REMANUFACTURING

Xerox and CopyMagic are in the minority in industrial remanufacturing since less than 5 per cent of remanufacturing in the USA is done by OEMs (Guide 2000). Additionally, both Xerox and CopyMagic rely heavily on leasing to enable the firms to forecast the timing and quantities of product returns. Product returns from leasing, for non-OEM remanufacturers, represents less than 5 per cent of total returns and this makes the tasks of forecasting product returns timing and quantities much more difficult. This forecasting problem manifests itself as product imbalances since matching return rates with demand rates is complex and difficult. OEMs have another distinct advantage in the area of product design, since most products must be designed for re-use: a modular design and clearly labeled materials. The volume of returns in the case of value-added remanufacturing is significantly less than for container re-use.

Marketing is more complex for the remanufactured products since customers may require significant education and assurances to convince them to purchase remanufactured products. Market cannibalization is also a significant concern since little is known about how remanufacturing sales affect new products sales. The characteristics of this supply chain are listed in Table 40.2.

Table 40.2 Characteristics of closed-loop supply chains for industrial remanufacturing

High variances
Stable production technology
Limited volumes
Modular design
Imbalances in supply and demand
Cannibalization

CLOSED-LOOP SUPPLY CHAINS FOR THE RE-USE OF CONSUMER ELECTRONICS

Our last case discusses the re-use of consumer electronics equipment and details the supply chain and remanufacturing operations.

Cellular Telephone Re-use at ReCellular, Inc.

ReCellular, Inc. was founded in 1991 in Ann Arbor, Michigan by Charles Newman to trade new, used and remanufactured cellular handsets. The business grew from a venture that provided cellular telephones for leasing, and alternative sources for handsets were required to reduce costs (hence the discovery of the used handset market). ReCellular is a trading operation that refurbishes cellular phones when necessary to add value for existing orders, and buys and sells wireless handsets of all technologies. In 2000 ReCellular estimated it had remanufactured over 1.3 million cellular phones. One of the goals of the company is to be the 'first in the second' in the wireless exchange plan. The company offers remanufactured (refurbished) products as a high-quality, cost-effective alternative to new cellular handsets. Customer services include grading and sorting, remanufacturing, repackaging, logistics, trading and product sourcing (all services are specific to cellular handsets and accessories). ReCellular operates globally with a presence in South America, the Far East, western Europe, Africa, the Middle East and North America. The company has plans to expand operations to provide better coverage throughout the world.

The cellular communications industry is a highly dynamic market where the demand for telephones changes daily. Demand may be influenced by the introduction of new technology (for example, digital and analog), price changes in cellular airtime, promotional campaigns, the opening of new markets, churn (customers leaving present airtime providers) and the number of new cellular telephones manufactured. Additionally, there is no worldwide standard technology (for example, Europe uses GSM, but the USA does not support this wireless technology) which necessitates dealing in a number of often disparate technologies and standards. These global differences make regional activities difficult since there may be no local market for certain types/models of phones, requiring a firm to manage global sales and procurement. Additionally, cellular airtime providers may limit the number of telephones supported by their system and the dropping of a phone model by a major carrier can greatly affect a local market. These factors make competition for an original equipment manufacturer challenging. However, a company offering used or remanufactured equipment faces numerous factors affecting the supply of used cellular phones. The same factors that complicate demand affect the availability of used handsets. Further, identifying and taking actions to reduce the uncertainties in the reverse flows as early as possible is an important issue in this business. The supply of used handsets is a volatile market, with volumes and prices in a constant state of flux. Supply uncertainty is not a complication faced by traditional OEMs.

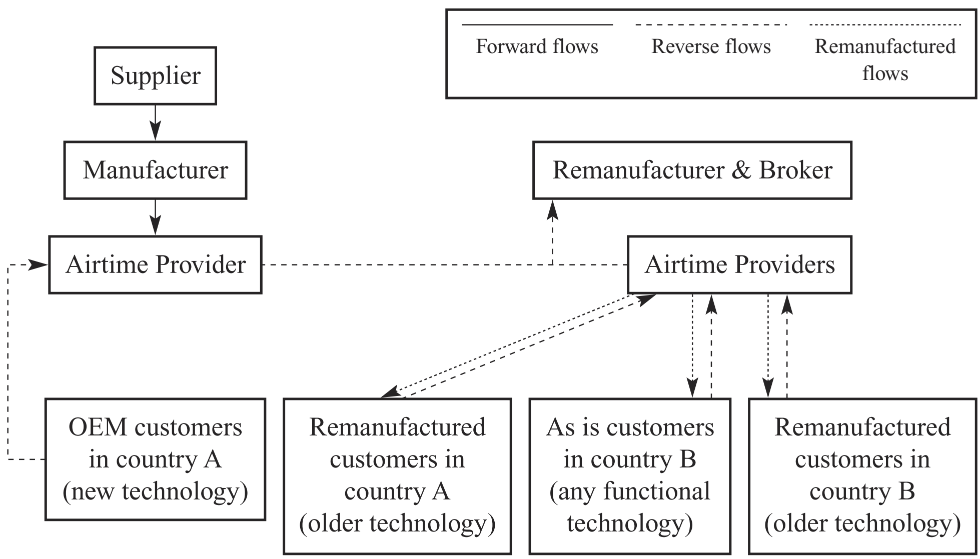


Figure 40.4 A closed-loop supply chain for cellular telephones

In order to fully understand the nature of the market, both forward and reverse flows of materials must be considered. Figure 40.4 shows the supply chain system for cellular telephone re-use. The forward movement of materials consists of the traditional flows from suppliers to manufacturers, manufacturers to airtime providers (retailers in this case, since the sale of a cellular phone is tied to airtime activation) and airtime providers to the customers. The reverse flows are more complex. Remanufacturers of cellular telephones do not collect handsets directly from the end-user, but rather rely on airtime providers or a variety of third-party collectors (we discuss the specifics of product acquisition in the next section). Airtime providers and third-party collectors act as consolidators who then broker the units to remanufacturers. ReCellular then sorts and grades the handsets, and sells the handsets 'as is' or remanufactured to airtime providers and third-party dealers working with airtime providers. Some handsets may be obsolete or damaged beyond higher-order recovery and these products are sold to scrap dealers and recyclers (note that this flow may come from both remanufacturers and third-party collectors). Recyclers recover polymers and other materials in the handset assemblies, and base materials in batteries. Scrap dealers may separate the handsets into materials and resell individual parts for re-use in other applications and offer the other sorted materials to recyclers. Suppliers may then purchase the recycled materials for use in new products.

The acquisition of used telephones is central to the success of a remanufacturing firm. The nature of product acquisitions is driven by what future demands (unknown) will be for phones. The lead times for delivery after used phones have been purchased are often lengthy and subject to a large amount of variability. This has caused remanufacturers to have stocks on hand to compete for sales. ReCellular obtains used phones in bulk from a variety of sources, including cellular airtime providers and third-party collectors. Third-party collectors are often charitable foundations (for example, the Wireless Foundation: *http://www.wirelessfoundation.org*) that act as consolidators by collecting used handsets and accessories from individuals. Cellular airtime providers also act as consolidators by collecting used phones from customers who have returned the phones at the end of service agreements, or customers upgrading to newer technology. Both these and other sources worldwide may offer a variety of handsets and accessories in varying condition for a wide range of prices and quantities. Owing to the low cost (approximately $0.50 per phone using air transport) of bulk transport of phones, using a worldwide network of suppliers of used phones is practical and cost-efficient. No individual returns are accepted since the channels required for direct returns from the consumer have too high a cost to be effective at the present time. Obtaining the best grade of used products for the best price is one of the key tasks necessary for the success of ReCellular. Deciding on a fair price to offer for the used phones is a difficult and complex task. At present, the acquisition staff devotes much of its time to identifying reliable and reputable sources of used phones and establishing a working relation with these suppliers. New suppliers usually require a visit in person to ensure the quality of the used phones.

The value of a used handset is highly dependant on future market demand for that particular model either in remanufactured or 'as is' form. The present demand for a graded 'as is' used cellular phone or a remanufactured phone is known for that instant in time, but, owing to the highly dynamic nature of the industry, these prices are not stable. The market forces discussed earlier may cause the value of a particular model of phone to drop or rise with little warning. An additional factor is that the selling price for remanufactured phones tends to drop over time, making the used phones a perishable product.

This nature of the product re-use market necessitates a fast, responsive supply chain that identifies sources of used phones for a fair price, and future buyers of these phones in either graded 'as is' or remanufactured condition. Additionally, the system must procure the phones in a timely manner, sort and grade the telephones, have the capability to remanufacture the phones rapidly to order and provide a fast accurate transport method to ensure timely delivery of the phones. ReCellular is developing an e-commerce site strictly for business to business in order to facilitate matching suppliers and buyers of equipment. The present e-commerce site shows the current stocks (model, price, grade and quantity) available and what models are needed (but not the prices offered). The site is being upgraded to allow real-time transactions by sales and procurement agents. Future considerations also include using the e-commerce site to facilitate on-line auctions for used and remanufactured products.

CHARACTERISTICS OF CLOSED-LOOP SUPPLY CHAINS FOR CONSUMER ELECTRONICS RE-USE

The volume of cellular telephones in use worldwide is enormous, with over 55 million cellular subscribers in the USA alone (US Central Intelligence Agency 2000). One of the first requirements for a remanufacturer in this environment is global coverage. Since the rate of technology diffusion is different for each country in the world, phones which may be technically obsolete in Norway may be current technology in Ecuador. This imbalance in the diffusion of technology makes having global operations and intelligence crucial for a profitable operation.

Tightly tied to the concept of global markets is the problem of acquisition, or obtaining the best-used product for the best price. The prices for cellular telephones are highly dynamic and are based on future expected prices for remanufactured handsets. The problem is further complicated by the market for graded, 'as is' cellular telephones where a selling is price is known, as opposed to an expected price in the future for a remanufactured item.

Cellular telephones are perishable items because of the high clockspeed, the time between new product or model introductions, in new product development. Electronics industries have the highest clockspeed, an average of 18 months, and this makes responsive systems crucial to making a profit. Remanufacturers cannot afford to remanufacture to stock in this environment since the value of the items drops daily. The characteristics of this supply chain are listed in Table 40.3.

Table 40.3 Characteristics of supply chains for consumer electronics re-use

Dynamic spot markets for supply and demand
High volumes
Perishable good
Cascade re-use opportunities (worldwide market)
High information requirements
High variability

MANAGEMENT OF CLOSED-LOOP SUPPLY CHAINS

In the simplest terms, all closed-loop chains do have a common set of activities. The recovery process consists of several highly interrelated sub-processes: product acquisition, reverse logistics, inspection and disposition (consisting of test, sort and grade), reconditioning (which may include remanufacturing) and distribution and selling of the recovered products. However, the previous cases illustrate that, while there are common processes, not all closed-loop supply chains are alike. Each different supply chain system has different characteristics and management concerns. Table 40.4 highlights the differences in the three forms of closed-loop supply chains we have discussed in this chapter. In the following sub-sections we discuss each of the sub-processes for the different types of closed-loop supply chains.

Table 40.4 Key distinctions between closed-loop supply chains

	Product acquisition	Reverse logistics	Test sort grade	Recondition	Distribution and selling
Refillable containers toner cartridges	easy	easy	easy	easy	easy
Industrial remanufacturing copiers	intermediate	hard	hard	hard	hard
Consumer electronics re-use cellular phones	hard	intermediate	easy	easy	intermediate

Refillable Container Closed-loop Supply Chains

'Product acquisition' refers to product acquisition management, and is actually a number of related processes (Guide and van Wassenhove 2000). First, product acquisition management determines whether re-use is a value creation activity for a specific firm. Second, if re-use activities are profitable, to maximize revenue the appropriate method for managing product returns should be selected. Third, operational issues, such as facility design, product planning and control policies, and inventory policies are dependent on the method selected to manage product returns. Fourth, product acquisition management activities help identify and develop new markets for re-used products, and to balance the return rates with market demands. We are concerned here primarily with the second and fourth product acquisition processes: selection of the appropriate method for managing product returns, and balancing return rates with market demands.

In the case of single-use cameras, the OEM controls the returns process by using cash incentives to motivate the photofinisher to return the empty cameras to Kodak's re-use facility. Xerox also directly controls the returns process for print/toner cartridges. Xerox supplies each customer with a pre-paid mailer and appeals to customers to send back the spent cartridge. Both strategies are extremely successful at ensuring constant volumes of containers, and are examples of market-based returns strategies. Market-based strategies are active strategies to encourage product returns, in contrast to waste stream systems where returns are passively accepted from the waste stream (Guide and van Wassenhove 2000). Both firms enjoy stable returns flows with predictable volumes each period.

Balancing return and demand rates in a refillable container closed-loop supply chain is a relatively easy process, in part because the customer cannot differentiate between re-used and new products. We do not mean that the processes involved in balancing return and demand rates are simplistic (see Toktay *et al.* 2000 for a complete discussion of this process), but rather that the process is simple in comparison with the other types of closed-loop supply chains. The technology contained in these products is stable and there are very limited secondary markets available for the used containers. There are secondary markets for the refilling of printer cartridges and single-use cameras, but these are small, localized operations that are considered more of a nuisance since the remanufacturing may be sub-standard and damage the firm's reputation. The returned products are mixed with new (replacement) materials as needed and then repackaged and sent back out through traditional distribution channels. There is no need to segment demand by product type (remanufactured v. new) or for a manufacturer to consider market cannibalization.

Reverse logistics activities are the processes required to move the products from the end-user to the facility where re-use activities will take place. In both examples of refillable containers, these sets of activities are simple. The photofinisher acts as a consolidator for Kodak and eliminates the need for Kodak to deal with individual customers. Xerox has minimized their contact with end-users by using a pre-paid mailer, which the customer may then use to arrange for pick-up and transport. Fleischmann (2000) provides a complete discussion of reverse logistics networks and their characterization.

The disassembly, test and inspection processes and the remanufacturing processes for refillable containers are simple. The product itself is simple and the costs are low; products that may be questionable may be recycled with little or no concern about replacement materials. Finally, the distribution and selling processes are simple since traditional distribution networks are used and the customer base is the same as for new products.

Industrial Remanufacturing Supply Chains

We summarize the key factors for success in Table 40.5. Of the closed-loop supply chains, the industrial remanufacturing sector is most likely the best documented, but the most difficult to plan, manage and control.

Table 40.5 Keys to success: industrial remanufacturing closed-loop supply chains

1. Ability to forecast and control the timing, quantity and quality of product returns (information systems [IS])
2. Design for re-use (recycle, repair and remanufacture) (engineering)
3. Customer education (remanufactured as good as new)
4. New relationships with suppliers (fewer parts & components, and design)
5. Complex production planning and control problems

Product acquisition activities for industrial remanufacturing may be based on leasing, in which case either the product will be returned at the expiration of the lease, or the lease will be renewed with the same item. Product acquisition in the case of leasing may be viewed as relatively simple; however, only a small number (5 per cent) of industrial remanufacturers report using leasing (Guide 2000). Remanufacturers report using a number of other techniques (deposits, rebates and cash refunds) with varying degrees of success and this indicates that the problem of obtaining sufficient quantities is more difficult than for refillable containers.

Balancing return and demand rates for industrial remanufacturing is more difficult since there are distinct and separate markets for new and remanufactured goods. Customers may require the newest technology or may simply perceive remanufactured products as inferior. These separate markets may affect the distribution and selling of remanufactured products. Market cannibalization may be a concern for manufacturers providing remanufactured products and, as a result, the retail markets may be disparate.

The reverse logistics process is most often very difficult, since the remanufacturer must arrange for pick-ups from many geographically diverse facilities. Many used industrial items are also regarded as hazardous waste and must be treated as such during transport. The process of testing, sorting, grading and inspecting is time-consuming and complex, with the possibility of a single product containing tens of thousands of parts and components. The screening process must be rigorous since products of poor quality may be expensive to remanufacture or dangerous to re-use. The remanufacturing/reconditioning processes are most often complex and difficult to plan, manage and control (Guide 2000).

Consumer Electronics Re-use

These types of closed-loop supply chains may hold the greatest promise, owing to the volume of products available for re-use, but at the same time these types of supply chains represent some of the greatest challenges. We list the key factors for success in Table 40.6.

Table 40.6 Keys to success: consumer electronics closed-loop supply chains

1. Ability to forecast and control the timing, quantity and quality of product returns in a global market (IS)
2. Fast response (perishable items)
3. e-commerce to identify buyers and sellers
4. Identify and exploit cascade re-use
5. Identify and exploit technology diffusion differences

Product acquisition is very hard for this form of closed-loop supply chains. The products are used globally, but the rate of technical diffusion is different in various geographic areas. This requires that a successful operation should have worldwide collection and distribution markets and these markets will not be in the same geographic areas. Supply and demand rates and prices are extremely volatile. The products are also perishable items, since the value of a remanufactured item may drop daily because of the rapid rate of technological progress and the rate of technology diffusion. There are also multiple options for re-use, since products may be sold in graded 'as is' condition or remanufactured. Each option has a different selling price which is quite dynamic.

However, there are several of the major processes that may be characterized as easy to intermediate. The nature of the products, with very few mechanical parts, makes them simple to test, sort and grade, and to remanufacture or recondition. The reverse logistics processes are somewhat hard to coordinate since there are so many national borders with customs regulations to manage. However, the handsets are small and light and may be shipped in bulk with commercial air carriers inexpensively (approximately $0.50 as of the summer of 2000). The distribution and selling processes involve a number of different nations, and require knowledge of the cellular technology in use and the airtime providers. The selling process is, for reasons discussed previously, tightly intertwined with the acquisition process.

CONCLUSIONS AND RESEARCH ISSUES

There are a number of unique structures for closed-loop supply chains. These different structures all require a set of common activities: product acquisition, reverse logistics, test, sort and grade, remanufacturing/reconditioning, and distribution and selling. The successful management of the various activities does not always involve the same actions from supply chain to supply chain. It is crucial for managers and researchers to understand the differences and the implications of these differences. One of the basic research needs is a continuing refinement of the documentation, categorization and understanding of the different forms of closed-loop supply chains. Other pressing research needs are those activities shown in Table 40.4 as 'hard'. Finally, we hope that holistic models will be developed that reflect the complex nature of the interactions.

41. Remanufacturing cases and state of the art

Geraldo Ferrer and V. Daniel R. Guide Jr

Remanufacturing is an environmentally and economically sound process to achieve many of the goals of sustainable development. Remanufacturing closes the materials cycle, and provides the basis for supply chains for product recovery and re-use. It focuses on value-added recovery, rather than just materials recovery, that is, recycling. By one estimate, there are more than 73 000 firms engaged in remanufacturing in the USA directly employing over 350 000 people (Lund 1983). Total sales are in excess of \$53 billion per year (USEPA 1997c). The US Environmental Protection Agency (EPA) cites remanufacturing as an integral foundation of re-use activities for its lower energy consumption and reduced waste generation. Remanufacturing has been described as

> . . . an industrial process in which worn out products are restored to like-new condition. Through a series of industrial processes in a factory environment, a discarded product is completely disassembled. Useable parts are cleaned, refurbished, and put into inventory. Then the new product is reassembled from the old and, where necessary, new parts to produce a unit fully equivalent – and sometimes superior – in performance and expected lifetime to the original new product. (Lund 1983)

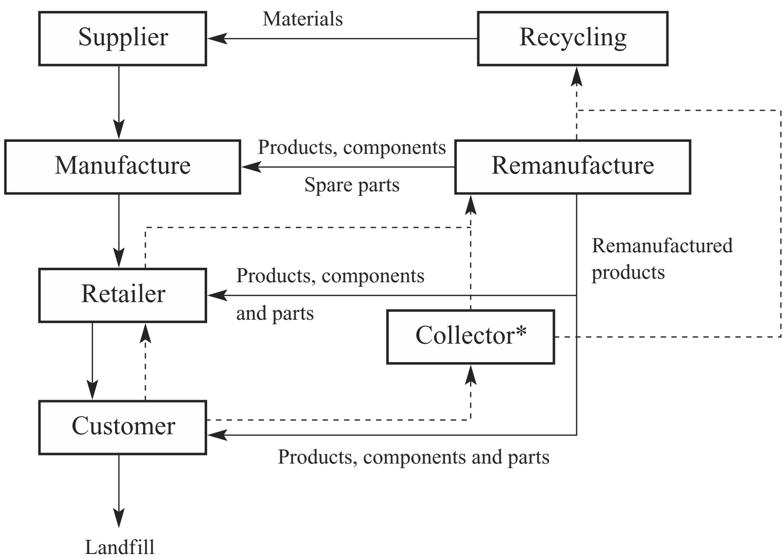
It is distinctly different from repair operations, since a product is completely disassembled and all parts are returned to as new condition, which may require cosmetic operations (further differences are discussed fully in later sections).

INDUSTRIAL ECOLOGY AND REMANUFACTURING

Industrial ecologists recognize that the industrial environment is an ecosystem that is not particularly efficient. The major problems are those of toxic waste and the inefficient use of materials. A great deal of the work in industrial ecology has concerned the development of technical solutions to problems associated with industrial processes rather than the development of managerial techniques. Jelinski *et al.* (1992) envisioned the industrial system as a closed environment in which by-products from some industrial processes could become inputs for other industrial processes. Frosch (1992) proposed the creation of closed-loop industrial systems since natural resources are limited and any closed ecosystem, that is the earth, can accommodate only a finite amount of waste. Piasecki (1992) called for the development of innovative management techniques designed to reduce waste, rather than reliance on technology. These concerns are freshly reviewed and greatly expanded in other sections of this handbook.

Supply chains incorporating product recovery and re-use offer potential advantages to corporations, including increased profitability through reduced materials requirements and improved market share based on environmental image (Ayres, Ferrer and van Leynseele 1997). These supply chains seek to minimize material waste by recovering the maximum content of returned manufactured products. This approach has the advantage of minimizing the amount of materials being landfilled and providing as new units for only a fraction of the energy, materials and labor required to manufacture a new unit.

A supply chain with product recovery and re-use is composed of a number of highly dependent subsystems (see Figure 41.1). Firms have a number of options available for returning products to consumers in useable condition: product repair, remanufacture and complete materials recycling. The correct choice among these options may be dictated by economics and by the condition and age of the returned product. A supply chain designed for product recovery and re-use is fed with discarded items in place of virgin materials.



Note: * In the case of overstocks and warranty returns, the retailer may act as the collector.

Figure 41.1 The supply chain with forward and backward flows

In Table 41.1 we present some of the key distinctions between repair, remanufacturing and recycling operations. Product identity refers to whether or not a unit must be regarded as a whole, or simply the sum of interchangeable components and parts. The degree of disassembly is the level to which the units must be broken down in order to be recovered (the maximum being material separation for recycling). Extent of transformation is how much the end unit is changed as a result of the operations required. This transformation includes a broad spectrum of possibilities where a repair item will simply be returned to the consumer as the same product, a remanufactured item may include technical upgrades, and a recycled material is returned to the consumer in the form of a new manufactured product. Since both material and labor contribute by adding value in the transformation process, distinct economic measures are used. Remanufacturing operations also provide materials for recycling. Returned products may be technologically obsolete,

Table 41.1 Factors differentiating repair, remanufacturing and recycling

Operations	Product identity	Degree of disassembly	Extent of transformation	Material value-added	Labor value-added
Repair	unit	diagnostic	none	replace or repair defective parts	moderate
Remanufacturing	unit, component or part	complete	limited	replace unrecoverable parts, technical upgrades	extensive
Recycling	none	destructive	complete	none added	limited

or not economically remanufacturable. These types of returned products are candidates for recycling, since all the value remaining is in the materials themselves.

Repair processes normally seek to return a unit to useable condition by simply isolating the defective or broken component or part and replacing it. In this case, disassembly operations are limited to the minimum required to execute the repair(s), and the unit retains its identity, given by its serial number. The major sources of uncertainty in a repair environment are replacement materials and the operations required to make a unit useable. Materials planning in a repair environment is difficult since the usage rates for components and parts is stochastic and changing over time. The likelihood that a component or part will require replacement is a function of a number of interrelated factors including, but not limited to, age of the unit, operating conditions, whether the unit was designed with repair in mind, and the nature of the fault causing failure of the unit. These stochastic replacement rates make planning for materials an important area of concern, particularly when the cost of replacement components and/or parts is high. Also of interest is the strategic use of inventory to shorten repair lead times and the financial trade-offs involved.

Scheduling repair events and related production planning and control functions is complicated because the exact operations required to repair a unit are unknown until the cause of the failure has been determined. In the repair facilities that we have visited, most operations routings are limited to a closed set of possibilities that include the tasks required to repair the recovered product in the worst-case scenario. The routing/scheduling function may be further complicated by economic or material availability considerations that recommend replacement (based on expected cost) of a component or parts after a diagnostic inspection has been performed or mandate repair owing to the lack of suitable replacement parts. The stochastic nature of the operations required complicates the planning of capacity and setting delivery dates. There is a need for production planning and control systems that cope with the additional amounts of variability inherent in repair operations.

REMANUFACTURING STATE OF THE ART

We refer the reader to Guide *et al.* (2000) for complete literature reviews. We note that the majority of previous research concerned with remanufacturing focuses on a particular functional activity, such as network design, shop floor control or inventory management. The research literature has a number of restrictive assumptions. First, many of the models assume that product returns are an exogenous process. Second, when product returns are explicitly considered, return rates are assumed independent of sales rates. Third, return rates are assumed to be outside the control of the firm. Finally, although several authors have called for a more integrated approach to logistics planning, very little research has addressed such integration.

Under the typical assumption of exogenous return rates and uncontrollable quality, there is research showing that remanufacturing operations are more complex to plan, manage and control than traditional manufacturing (Guide and Srivastava 1998; Guide, Srivastava and Kraus 1997). The primary reason for this complexity is the high degree of variability in the quality of used products that serve as raw materials for the production process. Since the condition of used products may vary widely, the tasks of materials planning, capacity planning, scheduling and inventory management are complex and difficult to manage. Managing this high amount of variability is expensive for the firm, since decoupling the system requires higher investments in materials, equipment and labor.

Ferrer and Whybark (2000) describe a fully integrated material planning system to facilitate managing a remanufacturing facility subject to parts commonality and uncertainty in core returns. The system is based on materials requirements planning logic, something that many firms are already familiar with. They are concerned with developing systems to help manage the firms dedicated to remanufacturing individual car components for the replacement market.

THE ECONOMICS OF REMANUFACTURING

Intuitively, remanufacturing is good for the environment. It gives a second life to materials, parts, components and the value added within. They would otherwise be landfilled or, in the best-case scenario, destroyed for the recovery of the materials through recycling. An important question remains: is remanufacturing economically sound? Ferrer (1997a) showed that remanufacturing can be a very profitable operation. Firms are often encouraged to offer environmentally friendly products (for example, remanufactured products) as part of being a good citizen. However, this is an unrealistic expectation since a rational firm will only engage in profitable ventures, those that increase shareholder wealth. Further, it may not be reasonable for every original equipment manufacturer (OEM) to engage in re-use activities. The fastest growing firms in electronics and telecommunications may need all the available capital to invest in core activities as well as research and development. The stock market is expecting high returns from these sectors, and firms may require high returns on capital expenditures. It may be rational for an OEM to not engage in re-use activities, to subcontract the re-use operations, or to encourage start-up corporate spin-offs to assume the responsibility. The decision whether or not to engage in re-use activities directly, indirectly or not at all should be driven by a thorough costs and benefits analysis.

The Value of the Core in Remanufacturing

Here is an approach for analyzing the value of a core entering the remanufacturing plant. The used product (CV) should be valued as the market value of the remanufactured

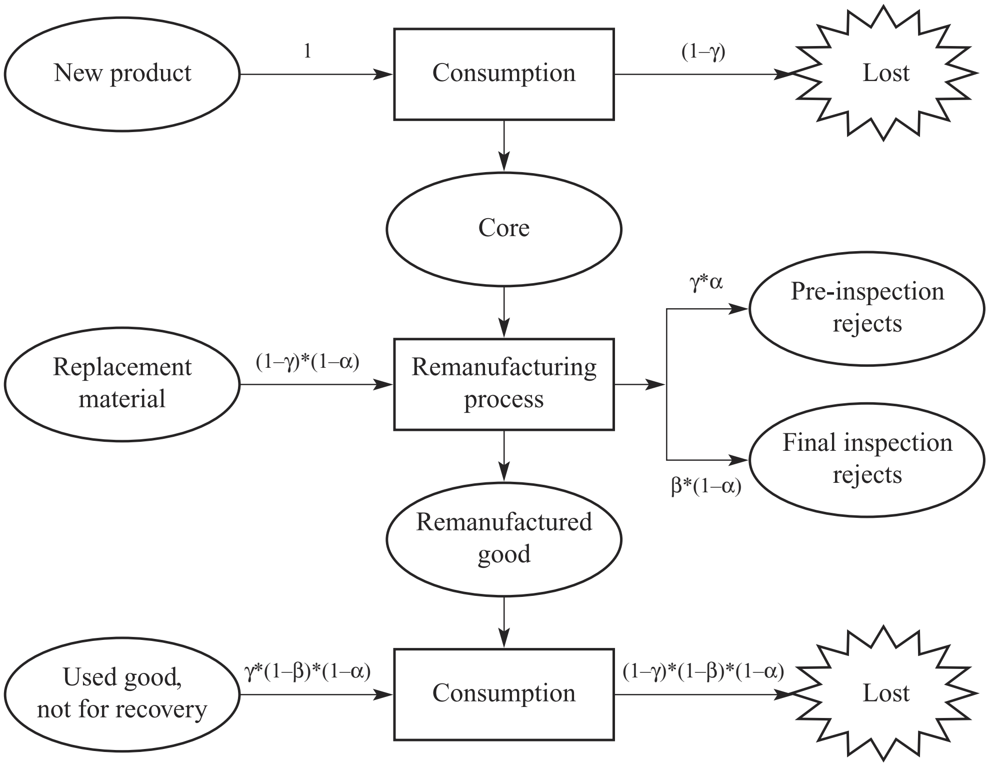


Figure 41.2 Material flow, single recovery

product (*MVRP*) minus the material cost (*VMC*) and the process cost (*PRC*), adjusted for two yields: the loss incurred in the disassembly process (Y) and the loss incurred in the inspection processes (Y_{insp}). That can be expressed as:

$$CV = MVRT * Y - (VMC + PRC)/Y_{insp}.$$

Figure 41.2 shows the material flow, assuming that the core is remanufactured just once, it has a second life and is finally disposed of at no cost. A fraction α of the cores is rejected during pre-inspection. At the end of the process, all remanufactured products are inspected and a fraction β is rejected. Hence,

$$Y = (1 - \alpha)(1 - \beta) \text{ and } Y_{insp} = (1 - \beta).$$

A core returning to the remanufacturing facility provides an opportunity value, given by *CV*, which is exploited when it is effectively remanufactured. This expression should be used with caution, because of the assumptions that directly affect the result. It overlooks the transport cost to collect cores and the revenue from the sale of recyclable materials and from tipping fees. More important, though, the operating yields are sensitive to both exogenous and endogenous causes. Not much can be done about the exogenous yield changes. However, as the remanufacturing technology progresses, the process yield is expected to improve.

Dynamic Analysis of Product Recovery Processes

Figure 41.3 balances the material flow, assuming that each product can be remanufactured indefinitely, without reduction in the length of product life. Some units are lost and never returned. If both the pre-inspection reject and the final inspection reject equal 5 per cent, the system is in equilibrium if, for each 100 units demanded in the market, 90.25 are remanufactured and 9.75 are new. If each unit is allowed to be remanufactured an unlimited number of times until they are rejected, the expected number of recovery per unit equals 90.25/9.75. Hence, on average, each core is remanufactured approximately 9.25 times before being eventually rejected.

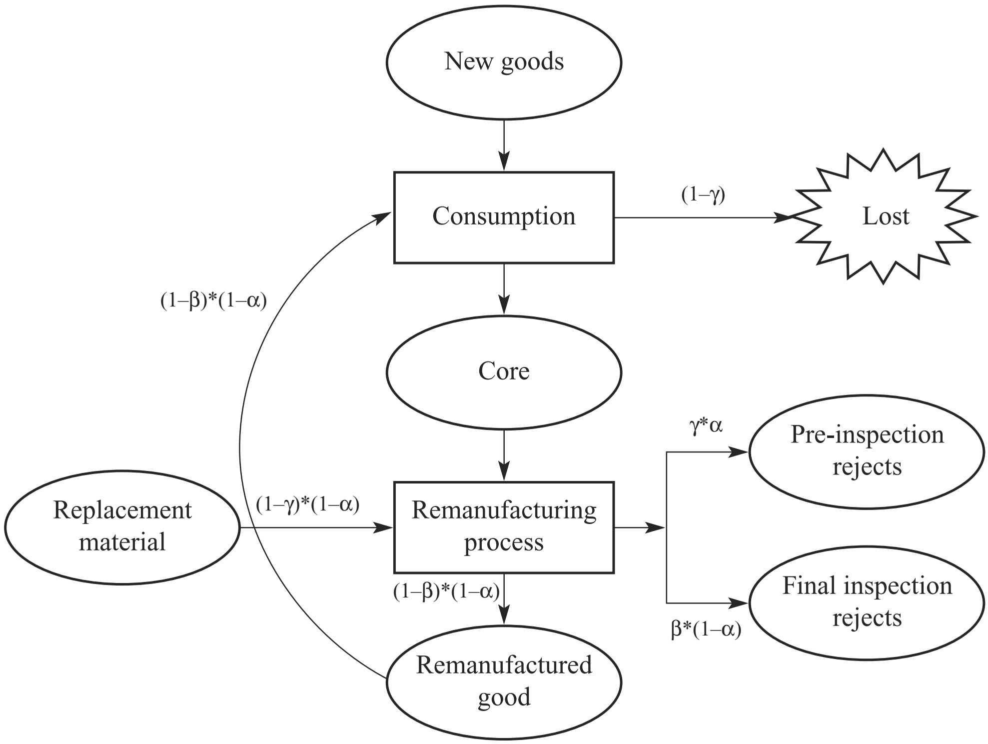


Figure 41.3 Material flow, multiple recovery cycles

If n is the maximum number of recovery cycles that a core is allowed to go through, the expected revenue, $ER(n)$, is given by the value generated by the original product (NPR) added to the revenue generated by each remanufactured product (RR). Likewise, the expected cost, $EC(n)$, is given by the cost to acquire the new product (NCP) added to the cost to remanufacture in each cycle (RC). It is simple to show that the expected revenue and the expected cost per core is given by the expressions:

$$ER(n) = NPR + RR^* \, (Y - Y^{n+1}) \, / \, (1 - Y)$$

$$EC(n) = NPC + RC^* \, (Y - Y^{n+1}) \, / \, (1 - Y),$$

where $Y = (1 - \alpha)(1 - \beta)$is the total yield and $(1 - \gamma)$ is the proportion of the material that is dissipated in the environment and not returned to the recovery facility.

In practice, the core cannot be remanufactured more than a limited number of times. The pre-inspection rejection rate tends to increase with the number of times that the core is remanufactured, because of the fatigue of the core structure. In order to ensure that a large number of cores pass the pre-inspection process, practitioners reduce the cycle time as the number of cycles increase. Suppose that a core is remanufactured three times before it is finally disposed of. In the tire industry, it is common to use the rule-of-thumb that the length of the three lives should be in the proportion 1 : 0.8 : 0.7. If the cycle time is reduced, one should expect that its value is reduced in the same proportion. Hence, for analytical convenience, we propose a recursive rule for estimating the life length of a remanufactured product to maintain a constant acceptance rate:

$$RR_n = Z * RR_{n-1}$$
$$RR_0 = NPR,$$

where Z is the rate of reduction of the cycle time. Under this assumption, the expected revenue becomes:

$$ER(n) = RR_0 + Y \times RR_1 + \ldots + Y^n \times RR_n$$

$$\sum_{i=0}^{n} = Y^i RR_i = NPR \times \sum_{i=0}^{n} Z^i Y^i$$

$$ER(n) = NPR \times \frac{1 - Y^{n+1} Z^{n+1}}{1 - YZ}$$

Notice that, for each additional cycle, one should compare the expected additional revenue with the expected additional cost. The optimal policy allows remanufacturing up to n times if and only if the expected additional cost of remanufacturing the nth time is less than the expected additional revenue. Moreover, the expected additional cost of remanufacturing the $(n+1)$th time is greater than the expected additional revenue. This translates to

$$Y^n Z^n NPR > Y^n RC$$
$$Y^{n+1} Z^{n+1} NPR < Y^{n+1} RC.$$

The solution to this is given by the integer n where

$$n = \left[\frac{\log(RC/NPR)}{\log(Z)} \right].$$

Applying typical values of RC, NPR and Z, the recommended number of recoveries for each core varies from two to six. The result greatly depends on the nature of the product and of the remanufacturing market at hand. For a comprehensive application of this model, see Ferrer (1997b).

SUPPLY CHAINS WITH REMANUFACTURING

The impact of product design on the potential for later product re-use cannot be overestimated. There are a number of considerations for post-consumer product recovery and there are challenges associated with them. However, many products are currently 'designed for disposal' because the product designers are not aware of the impact their designs have on post-consumer re-use. Worse yet, in some industries there have been efforts to prevent higher order re-use of products. Mobile telephone manufacturers may lock the phone, via ROM programming and cryptographic techniques, to a particular airtime provider. This practice of locking makes remanufacturing mobile telephones virtually impossible, unless the manufacturing firm is willing to provide the software keys.

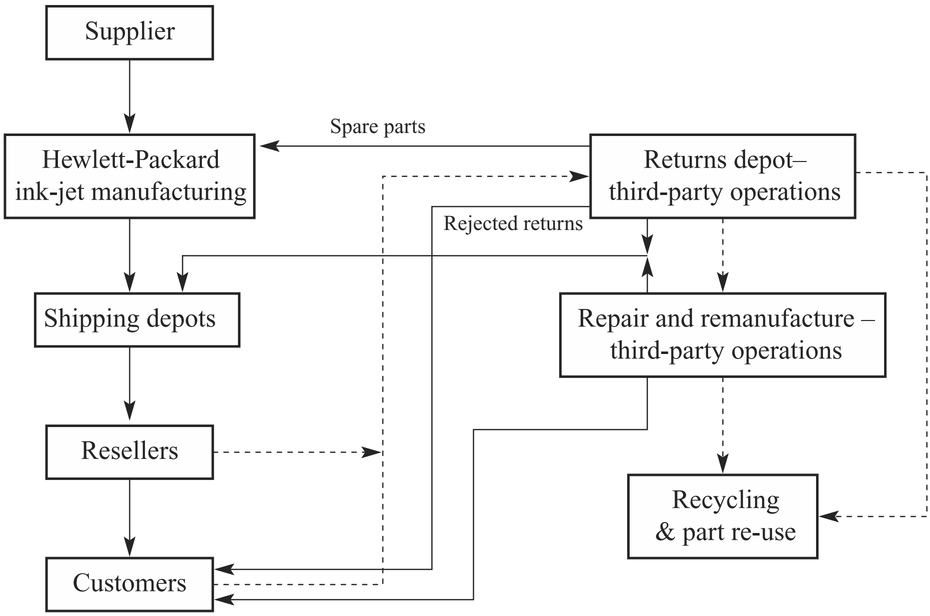
For the most part, designers of traditional supply chains do not consider the reverse flows and re-use of goods either. However, there are many instances of product returns that are not motivated by industrial ecology concerns. Consumers routinely return products for warranty service, product upgrades or because of dissatisfaction, and many resellers are allowed to return overstocks of products. Some firms have a formal strategy to manage return flows, but most firms accept product returns as a cost of doing business. They view returns as a nuisance and treat the activity as a cost center. In what follows, we describe a case of product re-use.

Reverse Logistics at Hewlett-Packard Company

Reverse logistics at Hewlett-Packard's ink-jet printer division are driven by product returns from end-users and resellers. HP does have environmental programs, but they are focused on materials selections for ease of recycling, minimizing energy consumption and other materials-focused re-use alternatives. The company contracts with third-party vendors for product returns and remanufacturing, but the activities, until recently, have been considered a cost center, rather than a source of profits.

The product returns process is part of the Hewlett-Packard supply chain system where products are returned from resellers and end-users (see Figure 41.4). Customers may return printers for a variety of reasons, including incorrect knowledge of the characteristics of the printer purchased (printing quality, speed, compatibility and so on), remorse, or because the printer will not function properly (manufacturing defect). Resellers also return products because of overstocks and consumer returns. After the returns are received at the depot, a visual inspection is done to ensure that the box contains the proper contents. If the product return is in an unopened carton, the units are overstocks and may be repackaged, and transported to the main HP distribution warehouse. A credit must be issued to the reseller for the overstocked units returned. HP does charge a fee for processing product returns, and these small fees do little to offset the reverse logistics costs. These restocking costs are set by the market and are considered a cost of doing business in the consumer electronics market.

If a product is returned in an opened carton, HP must treat the product as a defective



Note: Dashed lines for reverse flows.

Figure 41.4 Forward and reverse product flows for HP ink-jet printers

item and assign the unit to testing and refurbishment (remanufacturing). Individual parts and components (and in some cases entire units) may be used as part of the spare parts network maintained by HP. HP views sales of remanufactured printers as an opportunistic activity that may cannibalize sales of new printers. (HP estimates that, for every four refurbished units sold, a new sale is cannibalized. It should be noted that this is a very subjective estimate and that no consumer studies were done to arrive at this rule of thumb.) A third-party provider does all remanufacturing of the printers for Hewlett-Packard. However, a significant number of items (more than 10 per cent) are tested and found to be new items that had had the carton opened by a reseller to avoid restocking fees. These items are returned to finished good inventory. The lead time required to return products, sort and provide for final disposition is quite lengthy considering the short life cycle for consumer electronics.

The product development cycle for printers is growing shorter and this places additional pressure on the product returns process. Since many products are returned toward the end of the product life cycle, there are fewer opportunities for HP to recover the full value of the product. Additionally, because of the lengthy cycle times required for carrier pick-ups (products may sit for extended periods of time) products may be delayed from disposition until later in their life cycle.

It is rational that the reverse logistics and remanufacturing processes are designed to be as effective as possible. First, the cycle time required to identify, transport and make dispositions must be minimized in order to provide HP with the most options for product re-use. The goal is to have a returns facility where items are unloaded, entered into the system and processed (rebox, repair, remanufacture or scrap) without interruption. The present operation requires that the products wait at a number of steps as the boxes are open, resealed, reopened and finally processed. Meanwhile, the products are stored in large inventory holding areas for extended periods.

Decision models are also being developed that aid decision makers as to the best use of returned units. These decisions include how many times a model should be remanufactured, and whether or not to remanufacture individual products and product lines. A program designed to develop new markets for recovered products is also in progress. HP is investigating partnerships with third-party providers where HP insures the quality of the remanufactured printers. This contractor is responsible for remanufacturing and marketing the recovered printers. Subcontracting is necessary because remanufacturing is significantly different from traditional manufacturing (consequently, outside HP's manufacturing competence) and because a detailed cost analysis showed that the margin from these types of operations could not meet HP's cost of capital hurdle.

The supply chain system must be responsive because of the short life cycles of printers. Each day a printer spends out of the reseller network, the value of the printer drops. The reverse materials flow is more difficult to manage because of the higher inherent uncertainty, and the need for timely and cost-efficient operations. HP has currently to maintain active control of the vendors for acceptable quality of remanufactured and repaired items. HP is currently investigating a number of initiatives to improve the re-usability of their printers and to provide a more responsive system for reverse logistics and disposition of product returns.

PRODUCT ACQUISITION

Guide and van Wassenhove (2000) classify the systems for obtaining used products from the end-users as *market-driven* or *waste stream*. The waste stream system relies on diverting discarded products from landfill by making producers responsible for the collection and recovery of their products. A market-driven system relies on financial incentives to motivate end-users to return their products to a firm specializing in the recovery of those products. In the waste stream system, firms passively accept all product returns from the waste stream. Unable to control the quality of returns, firms often consider the large volumes of returns a nuisance, and naturally tend to focus on the development of low-cost reverse logistics networks. In the European Union (EU) a number of recent legislative acts, known as 'producer responsibility laws', require manufacturers to assume responsibility for the end-of-life disposition of their products. Some legislative acts may require the end-user to return the used product to a collection system for re-use. The requirements for firms doing business in the EU are clear, and these regulations may act as entry barriers for firms not aware of the changes required for reverse logistics activities. The result of the product returns mandates and policies is a large, uncontrolled volume of used products flowing back in increasing volumes to the original equipment manufacturers. Firms are ill-prepared to cope with the complexity of product returns and end-of-life disposition, and are seeking ways to minimize their losses.

In a market-driven system, end-users are motivated to return end-of-life products by financial incentives, such as deposit systems, credit toward a new unit or cash paid for a specified level of quality. Firms are able to control the level of quality of returned products since acceptance of returns is conditioned by standards. Market-driven systems are common in the USA because of the profitability of remanufacturing (Guide 2000). Firms using a market-driven approach for product returns focus mainly on high-value industrial products. However, there are exceptions; the system for remanufacturing automotive parts has been well established since the 1920s. For instance, Hormozi (1997) reports that Henry Ford realized that valuable automotive components should not just be discarded, but should be rebuilt. So the Ford Motor Company authorized a few select dealerships to remanufacture replacement parts. Soon, Ford established a franchised network of remanufacturers, authorized to recover Ford components on a regional basis. That historical beginning was soon followed by other carmakers who realized that franchising a few remanufacturers was an efficient way to deliver replacement parts for their products. Automotive component remanufacturing is not confined to large firms, however. A number of small companies have been established to remanufacture components for car and truck maintenance needs.

A combination of the market-driven and waste stream approaches is also possible. Product returns may be mandated or encouraged by legislative acts, but firms may still encourage the returns of products in known condition by offering incentives. A firm using a pure waste stream approach or a pure market-driven approach will have facilities with different operational characteristics and managerial control problems resulting in extremely different views of re-use activities. The market-driven system views re-use as a profitable economic proposition. In the waste stream system, cost reduction is encouraged and the fundamental issue is to minimize the amount of money the firm spends on it. Past research has focused almost exclusively on the waste stream approach to re-use activities and it is logical that the modeling efforts are aimed at cost minimization.

CONCLUSIONS

There is a growing body of evidence that remanufacturing is an economically viable method to support a closed-loop materials cycle. Firms face a number of significant challenges in designing supply chains with reverse materials flows and in planning and controlling remanufacturing operations. There are also a number of technical issues such as product design, and the optimal number of times a component or part may be remanufactured, that desire consideration. The use of e-commerce as an enabling technology deserves close attention and documentation. Finally, there are many lessons available from commercially driven product returns that may provide valuable insights for the continuing development of industrial ecosystems.

42. Industrial ecology and extended producer responsibility

John Gertsakis, Nicola Morelli and Chris Ryan

As noted in other chapters of this handbook, green design or design for environment (DFE) and cleaner production can address an extensive list of environmental issues throughout a product's life cycle. Nevertheless, some impacts are currently beyond their control, especially those associated with discarded products. The bottleneck is often disposal, and it cannot be overemphasized that DFE features in a product can only facilitate – and not ensure – recycling.

A relatively new direction in government policy is now being adopted by most OECD (Organization of Economic Cooperation and Development) countries. It encourages manufacturers, in particular, to accept greater responsibility for their products when they reach end-of-life (EOL) and are discarded. Extended producer responsibility (EPR) represents a more systematic approach with the potential to revolutionize the way products are conceived, used, recovered and ultimately re-used, recycled or disposed of. The OECD has provided a definition of extended producer responsibility:

> EPR is defined, for the purposes of the OECD project, as the extension of the responsibilities of producers to the post-consumer stage of products' life cycles. EPR strategies suggest that the use and post-consumer phases of a product's life cycle are important aspects of the 'pollution' for which responsibility must be assumed under the Polluter Pays Principle. (OECD 1996b, pp. 15–16)

A key objective of EPR, given the OECD definition, is 'to transfer the costs of municipal waste management from local authorities to those actors [i.e. the producers] most able to influence the characteristics of products which can become problematic at the post-consumer stage: waste volume, toxicity, and recyclability' (ibid., p. 16). By transferring these costs, governments hope to provide powerful incentives for producers to prevent waste generation, reduce the use of potentially toxic inputs, design products that are easily recyclable and internalize the costs.

EPR is a logical extension of the 'polluter pays' principle. It rests on an argument that the environmental impacts of resource depletion, waste and pollution are a function of the system of production and consumption of goods and services. Those impacts are substantially determined at the point of production, where key choices are made – on materials, processing and finishing technology, product function and durability, systems of distribution, marketing and so on. If that system is to evolve in a way that reduces environmental impacts, there is a need for policies that create appropriate *feedback* mechanisms for producers that will direct their investment towards continuous environmental improvement. In brief, EPR can be considered as an effective policy mechanism to promote the integration of the life cycle environmental costs associated with products into the market price for the product. The transferral of waste management costs is seen as a critical driver or incentive for industry to internalize the complete range of costs associated with managing end-of-life waste.

Initially, EPR attracted much attention within the context of Germany's 'Ordinance on the Avoidance of Packaging Waste' passed in 1991. The aim was to reduce waste by shifting the costs of managing packaging waste from the public sector to private industry, the envisaged result being a more innovative and environmentally oriented approach to packaging design and recycling. Although the ordinance and its rapid introduction caused much distress in German industry, there are no signs that the application of EPR is lessening. Indeed, the application of EPR is being expanded to other product sectors such as computers and copiers, domestic appliances, carpets, furniture, power tools, rubber tires and mobile telephones.

Stimulating industry to accept responsibility for its products at their end of life is an important focus for environment and industry policy in Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, the Netherlands, Norway, Sweden, Switzerland and the UK. It also applies in Japan, Taiwan, Canada and in some US states. For instance, the Producer Responsibility Initiative (UK) is being 'specifically extended to a number of industry sectors, including packaging, tires, batteries, vehicles, electrical and electronic equipment' (Hicks 1994, p. 25).

Several OECD countries have legislation in place that provides for regulations, covenants, ordinances or other mechanisms to impose EPR for particular product categories (OECD 1996b, pp. 42–9). Product categories around 1995 included in, or intended for, EPR programs in OECD and EC (European Community) countries include packaging, tires, batteries, waste oil, CFCs, printed matter, electrical and electronic products, office equipment, cars, furniture, building products and agricultural plastics.

SHORT- AND LONG-TERM PERSPECTIVES OF EPR

A number of different policy tools are used or are under consideration by governments to implement EPR. These include voluntary agreements or covenants with industry to achieve waste targets, deposit refund schemes, product disposal charges and end-of-life product take-back requirements. Environmental labeling, environmentally based procurement programs and minimum recycled content requirements are also considered as part of the EPR policy armory (Davis 1995, p. 2).

Deposit Refund Schemes

Denmark, Switzerland and Sweden have required makers/suppliers of small consumer batteries to levy a refundable deposit and Austria operates a similar policy for fluorescent lights and tires. Korea has deposit refund schemes covering beverage containers, batteries, tires, televisions, washing machines and lubricating oils. Taiwan introduced a similar system for PET bottles in 1991 (OECD 1993b).

Advance Disposal Fees

An advance disposal fee is a fee set to a level estimated to cover real disposal–recovery costs, paid by the producer into a government fund. A variation on this approach is that consumers who handle the waste from a product in a certain specified way receive a refund of the fee at the end of the product's life. In Austria this approach is used for refrigerators and in Sweden it is used for cars. In 1993, the state of Florida in the USA imposed a 1 per cent fee on containers that had not achieved a 50 per cent recycling rate. This scheme raised over $44 million in its first year of operation (Boehm and Hunt 1995, pp. 12–13). Hawaii enacted a similar fee on glass containers. Some 21 US states have schemes for tires and 10 US states and most Canadian provinces have systems for beverage containers (Lifset 1995, pp. 37–51).

Voluntary Agreements/Covenants

The UK Producer Responsibility Act essentially commits the government to negotiate with various industry sectors to achieve a waste reduction strategy acceptable to, and managed by, the industry. The Netherlands covenant system sets voluntary agreements within a framework of long-term targets and provisions for enforceable ordinances or regulations should voluntary covenants fail to achieve the targets. A covenant on packaging wastes was the first result of the Netherlands approach; other agreements include one reached on cars, where the industry asked the government to introduce a fee to support EOL recycling. This agreement is binding on all parties in the production chain. For batteries, tires and agricultural plastics, however, the Netherlands government legislated after it was clear that the industry sectors could not agree upon a voluntary covenant.

Product Take-back Systems

As the OECD comments: 'the most rigorous and most recent EPR programs center around the product take-back concept, but are rarely framed so simply'. Product take-back is almost the paradigm of EPR; it involves the establishment of set targets for collection and recovery of products, with laws, regulations or ordinances setting out the responsibility of the individual producer or importer to 'take back and recover' their products unless certain other conditions are in place. Industry-wide schemes are generally allowed for. The best known of the product take-back systems relate to packaging, with the German Packaging Ordinance (1991) and its industry-wide scheme for collection and recycling – the Duales System Deutschland – being the most prominent and most studied. The German system is credited with reducing packaging waste (plastic waste fell from 923 thousand tonnes in 1991 to 823 thousand tonnes in 1995, as reported by Helmut Schnurer, the former German environment minister). It is now a significant source of new jobs, 18 000 having been created directly by the Duales System.

THE LONGER-TERM PERSPECTIVE

In a long-term perspective EPR could be used with the purpose of shifting the present production and consumption systems towards more efficient *metabolic* processes. Some studies based on the concept of industrial ecology advocate this approach as a more coherent and holistic means towards achieving sustainability. This perspective implies radical changes in the production and consumption systems and in manufacturers' mission. As will be noted below, certain companies have already proposed innovative strategies based on it.

The complexity of products' production and distribution systems suggested different strategies for different industrial sectors, but those strategies have the common objective to generate circular processes that allow for the re-use of products, components or materials. The initiatives proposed in this chapter can be divided into two different approaches. The first approach consists in the *internalization* of the procedures that allow for a better control over the products' life cycle. This is possible when companies retain the products' ownership and provide their clients with the *service value (of utilization)* of their products. The second approach consists in the creation of a symbiotic network of different companies, in which the value of end-of-life products and material is amplified, in order to make their recovery and renovation, remanufacturing or recycling easier.

INTERNALIZATION OF CONTROL PROCESSES OVER THE PRODUCT LIFE CYCLE

Companies that follow this approach retain products ownership and lease products to their customers; in fact these manufacturers are proposing a radical change in their mission by choosing to sell, no longer products, but *value of utilization*. This strategy would move the focus of manufacturers' activities from the production of *products* to the provision of *services*, whose efficiency would reinforce the links between producers and their customers.

By offering services instead of products, manufacturers would provide solutions that fit the clients' requirements. This change of perspective – from the customer's point of view – would shift the focus from the value of products to the value of their utilization. Furthermore, customers would have the possibility of choosing the most updated technology available for any products. As a result, manufacturers would have a vested interest in ensuring the highest possible efficiency of products and the utilization of products would be more intense.

Maintenance operations and selective resource recovery of components and faulty products is easier in this approach, because manufacturers control the product efficiency during the phase of use, maximizing the opportunities to keep products operating longer and for second or third lives, subject to reconditioning and performance upgrades.

THE CREATION OF A SYMBIOTIC NETWORK OF COMPANIES

This approach derives from the need to add value to the secondary materials and components emerging from the recycling process in industrial sectors in which the production and consumption system is quite complex. The value of secondary material depends on many factors such as the quantity of labor needed, the regularity of incoming and outgoing flows of products in the disassembly plants, standardization and innovation in products and of course the price of virgin material.

A systemic approach to the question of EPR suggests more integrated initiatives promoted by a network of industrial partners. In such a network the flows of material and information could be more regular and the value of secondary material could be stabilized. Furthermore, some initiatives described in the case studies are based on the possibility that the *effluents* from one category of products (that is, discarded products and components) be used as *resources* for other production/consumption processes. The integration between industrial companies may take either of two forms: (a) 'vertical integration' between companies in different production sectors involved in other phases of a product's life cycle (such as appliance manufacturers and recyclers) and (b) 'horizontal integration' between competing companies that cooperate in different geographical locations in order to enlarge take-back systems in new countries (for example, the alliance between different car companies in the automotive industry).

REAL WORLD RESPONSES IN EPR

The following examples provide a glimpse of the way some companies have adopted EPR as a means of both minimizing environmental problems and adding value to their products. Spanning several different manufacturing industries and product sectors, the case studies also demonstrate that product take-back is not just about compliance with environmental regulations or an obstacle presented by bureaucrats, but is also a driver for innovation and running businesses more efficiently and competitively.

New Business Development Resulting from EPR in the Furniture Industry

Two noteworthy US furniture companies selling their office furniture products around the world are Herman Miller Inc. and Steelcase Inc. Both companies have developed comprehensive environmental programs aimed at eliminating or minimizing life cycle environmental impacts, particularly those resulting in solid and hazardous wastes. A German furniture manufacturer, Wilkhahn, is also pushing the boundaries by developing ergonomic chairs with durability and recyclability in mind. The products manufactured by Herman Miller, Steelcase and Wilkhahn include office partitions, storage units, ergonomic chairs and workstations. Design for disassembly principles have been embodied in their furniture to help ensure that service and repair, end-of-life remanufacture and materials recycling are not only realized but add value to the overall enterprise.

The problems arise chiefly during manufacture and disposal, with little or no impact during the product use. And it is the disposal or end-of-life stage where solid waste becomes an issue, particularly when significant volumes of furniture are sent to landfill. From as early as 1984, Herman Miller has been practicing EPR primarily through its As NewProgram. Herman Miller not only takes back and remanufactures its furniture products, it has also established a subsidiary company, called Phoenix Design, to operate that business. One of the resulting products – AsNew – combines recovered components with new components, with the objective of providing more competitive products at a lower price (Herman Miller 1993).

Steelcase has also seen the value of taking a product back and keeping valuable components and materials out of the waste stream. Like Herman Miller, it has managed to establish a successful business division essentially based on waste avoidance through reuse. For several years Steelcase has, through a specialist subsidiary, offered its Revest system which purchases used Steelcase furniture, then refurbishes, refinishes, reupholsters and resells it (Steelcase 1993).

Wilkhahn offer their customers an integrated range of features and service that collectively address waste minimization without compromising product quality, functionality or performance. Although the complete range of Wilkhahn furniture aims to address environmental factors, its highest profile 'green' product is the Picto chair. The Picto is designed and built to minimize waste in accordance with the waste management hierarchy: reduce, re-use, recycle. As a primary objective, the Picto is 'built for longevity' with materials specified that help increase product durability and overall product life. Service and repair also plays a key role in helping keep Wilkhahn chairs out of the waste stream. Finally, when the chair is beyond repair or no longer required by the first customer, Wilkhahn will 'take back' the chair and either refurbish, re-use components or recycle materials.

EPR and the Role of Alliances in the US White Goods Sector

Over the last decade several states in the USA have introduced landfill bans on appliances. At the same time, the turnover rate for household appliances in North America has been increasing, with an estimated 40 per cent of appliances in North America due to be replaced. This has helped create a seemingly profitable market for the take-back and recycling of end-of-life white goods. In addition, many public authorities and utilities are acting on the need to remove older, less energy-efficient appliances from the marketplace as a means of reducing greenhouse gas emissions. This has also been a driver for the recovery, re-use and recycling of discarded appliances while simultaneously offering consumer incentives to purchase more energy- and water-efficient models.

Within the context of EPR, an interesting partnership approach has emerged in the USA. Companies like the Appliance Recycling Centers of America Inc. (ARCA) are now promoting joint initiatives with public authorities and with appliance manufacturers as a way of optimizing the efficiency of recycling infrastructure and extracting maximum value from the incoming flows of discarded white goods.

ARCA is a publicly held company that provides collection, recycling and re-use services for major household appliances. The company collects used appliances from new appliance retailers and other sources and reconditions a portion of these units for resale. ARCA has its own national network of stores, called 'ENCORE . . . the reused appliance store'. The scale of the network is now increasing as the quantity and availability of re-usable white goods increase. Appliances that cannot be sold in the stores are recycled by ARCA in an environmentally responsible manner. Most importantly, ARCA has developed a formal alliance with Whirlpool Corporation to provide environmentally sound appliance recycling services for used appliances that are scrapped when Whirlpool sells new replacement units to apartment complexes that are being refurbished. ARCA has agreed to use Whirlpool appliances to support the 'early retirement' programs sponsored by energy utilities that replace older household refrigerators with more energy-efficient models.

ARCA's business is expected to expand across North America because of the alliance with Whirlpool, providing ARCA with a reliable supply of used appliances. While the relationship with Whirlpool was meant to be exclusive for the initial partnership period, ARCA expects to form alliances with other white goods manufacturers as well.

Value Adding in the IT Sector through EPR

Technological advances are accelerating the innovation cycle with a resulting proliferation of IT products such as photocopiers. This process not only leads to the need for EPR policies and regulations, it also provides manufacturing companies with new opportunities for business development and more profitable customer relationships. Although not widespread, it is important to note that voluntary environmental initiatives focused on product take-back, remanufacturing and recycling preceded regulatory based responses to EPR. At a global level, the Xerox model has demonstrated that EPR need not be defined or driven by government policy.

Xerox's policy has for a long time been to anticipate regulations and to consider compliance with them as the bottom line for its environmental strategies. Since 1968, Xerox has been dismantling its copying machines and recycling parts. Xerox's approach to product recovery, remanufacture and leasing highlights an ideal scenario: that is, one where EPR has its foundations in the company's own sense of corporate responsibility and how it can be used to deliver environmental and economic benefits for all parties concerned. While often over-quoted, the Xerox story is an important example of EPR flourishing prior to, and in the absence of regulatory demands and targets. The company's approach to EPR also highlights the fact that manufactured products such as copiers are only one element in the life cycle management process. Xerox's corporate profiling and branding reinforces its self-declared status as 'the document company'. Xerox is now committed to providing not only products, but also all the 'services' needed for customers to copy documents. This conceptual shift in Xerox's mission is not only consistent with the company's progressive environmental approach, it also proactively addresses innovation in product development and customer service.

EPR at Xerox is embedded in the business process from the outset of customer relations through to the provision of full service maintenance agreements. Such an agreement offers a three-year 'total satisfaction guarantee' committing Xerox to replace the product with an identical or similar model free of charge if the user is not fully satisfied with it. The participation of users in the chain ensures improved re-use and recycling of supplies and used machine parts. In some countries Xerox is establishing alliances with other partners, in order to facilitate the take-back process. For instance, in Canada, the company has launched a partnership with Canada Post that encourages Xerox Canada customers to return reusable toner bottles, cartridge replacement units and other used Xerox copier parts by taking them to any of Canada Post's 19000 retail outlets nationwide. Xerox provides prepaid postal labels with all returnable units. Canada Post then delivers the units directly to the recycling operations set up by Xerox across North America.

In particular, the role of EPR at Xerox worldwide highlights the point that EPR is not totally dependent on government policy and regulation. More importantly, the Xerox story also shows that EPR is not, nor should be, an end in itself. The broader goal is environmental protection within a context of responsible business growth. Whether this occurs as a result of imposed regulations or of voluntary actions is not necessarily the critical or defining characteristic of EPR. Ensuring that the life cycle management process actually takes place is the imperative.

An Industrial Consortium as a Way of Managing EPR in the Automotive Sector

The car is one of the symbols of the modern age and therefore also becomes a symbol of the environmental crisis faced by modern cities. For this reason environmentalists and public opinion have often focused on the environmental impact of the car, as an indicator of the impact of the whole socioeconomic system in western countries. Stricter environmental legislation, rising landfill prices and the public demand for environmentally improved products have forced growing attention on the recovery and recycling of end-of-life vehicles (ELVs).

At present, 10 million cars are scrapped in Europe each year, but only 80 per cent find their way into official or organized recycling centers, with the remaining 20 per cent being illegally dumped and abandoned. As a result, one of the main objectives of EPR regulations is to encourage a more standardized approach for the recovery and processing of ELVs. The turnover of cars is increasing, especially in countries like Japan, where the average life of a car is between four and seven years. Some governments are encouraging the replacement of old cars with new ones as a way of maintaining more fuel-efficient and less polluting fleets.

Approaching the implementation of EPR from a collective perspective is gaining momentum around the world as companies with specific product sectors consider the overall advantages of working in a cooperative manner. While Fiat, BMW, Renault and Rover commenced their recycling programs independently and in different European countries, in recent years they have combined efforts so that individual companies could ensure an efficient and effective recycling service for their ELVs across several European countries. This consortium approach to EPR represents yet another model by which companies are implementing an environmental program beyond the point of sale and warranties. The economies of scale associated with partnering increased the viability of recovering and recycling ELVs across national boundaries when operating in isolation might not prove possible, either commercially, logistically or culturally.

The consortium approach commenced with the development of agreements with car dismantlers, in order to operate a selective dismantling of end-of-life cars. Therefore the quantity of materials extracted from an ELV, and their quality, have to be measured on the basis of time and work needed to extract them. For this reason car manufacturers have set up pilot disassembly centers to study disassembly methods and provide dismantlers not only with encouragement but also with detailed information and knowledge about effective and efficient techniques. The producer's responsibility does not cease with simply collecting and dismantling ELVs. The original car manufacturers also collect materials and readdress them to their own production plants or to other companies which will treat them and re-use them as a raw material for bottles, carpets and so on. In their manufacturing facilities, companies use secondary plastics for parts with lower performance requirements. For example, Fiat re-uses plastics from bumpers for producing air ducts and other (often hidden) interior components in new models.

The establishment of pilot disassembly plants has been a vital first step in forming the consortium. The know-how acquired through these plants is then transmitted to key stakeholders such as dismantlers and vehicle designers, all of which streamlines the disassembly and recycling process. Thus the role of EPR in the car sector is to seek to influence directly new vehicle design.

CONCLUSIONS

EPR is a dynamic and evolving area of activity in terms both of government policy and of business practice. Definitions of EPR are in constant flux as governments and industry engage with the ways in which the life cycle environmental impacts of products and services can be eliminated or minimized within the context of pursuing more sustainable patterns of production and consumption that also enable responsible economic development.

What is becoming apparent is that different product sectors in different regions around the world are seeking to meet government expectations on EPR in different ways, noting that in many cases industry is seeking to minimize the extent to which any one company should take sole responsibility for EOL products – their preference being to distribute or share obligations along the supply chain and with consumers. Having said this, there are also a growing number of significant examples, such as those discussed above, where EPR as a principle or strategy has been addressed on a voluntary basis by innovative companies that have exploited the commercial benefits and potential of actively recovering and adding value to their products.

The other critical factor is the extent of interaction required between and among environmental management approaches. To talk of EPR in isolation from DFE, remanufacturing, supply chain management or LCA is to fail to recognize the interconnections between product design, environmental assessment and EOL processing methods, all of which are increasingly vital tools in maximizing the commercial viability of EPR schemes. This further reinforces the need for such tools and methods to work in harmony in pursuit of sustainability. Furthermore, it also highlights the relevance of industrial ecology as a way of meshing approaches in a more holistic manner by which resources can be used in a hyperefficient way.

Finally, while EPR is often discussed in terms of government regulations, the essence of such policy objectives relates to the way sustainable production and consumption might be realized through increased levels of waste avoidance and resource recovery. This highlights that many of the real challenges rest with industry and its ability to be just as innovative in 'unmaking' products as it is in 'making' them. Similarly, consumers will need to consider their role and obligations in 'de-purchasing' their end-of-life products.

43. Life cycle assessment as a management tool

Paolo Frankl*

Life cycle assessment (LCA) is potentially one of the most instructive management tools for gaining insight into product-related environmental impacts. It is considered by many as a complementary and more comprehensive tool with respect to other environmental management systems (EMS) for supporting an effective integration of environmental aspects in business and economy. However, strong disagreement still exists among analysts about the practical use of LCA in business: 'LCA is the environmental management tool of the 1990s . . . LCA will be seen as an integral part of the environmental tool-kit' (Jensen *et al.* 1997, p. 28), 'LCA procedures are too expensive and complicated, they could only seldom be used' (Arnold 1993), 'many methodological choices are required and a number of aspects still need to be worked out, potentially jeopardizing the credibility of the outcome of LCA studies. This may lead to a decreasing confidence in and use of LCA by industry and governmental institutions' (Wrisberg *et al.* 1997). However, despite still existing problems and barriers, the methodology and 'technique' of LCA have significantly improved during the last few years. Consensus has now been reached on a number of key issues, and several ISO norms have been published (see also Chapter 12). Meanwhile the adoption of LCA in business has grown and LCA is now used in many industry sectors.

The objective of this chapter is to describe the current state of the art in the use of LCA as a management tool and the outlook for the future. It focuses on the possible uses of LCA, the dynamic and learning aspects of LCA adoption patterns and the current level of diffusion in European industry. Finally, it discusses the issues still to be overcome and the main recommendations for a future proper and increasingly widespread use of LCA in management within the framework of industrial ecology.

MAIN USES AND CURRENT LEVEL OF APPLICATION

LCA can be used in many different ways in companies and several classification schemes have been proposed. Berkhout (1996) and Smith *et al.* (1998) link the uses to the position of the company in the product chain. So do Jensen *et al.* (1997), who also specify the level of detail in LCA (conceptual, simplified, detailed) related to each category. LCA applications make a distinction between internal and external uses with respect to the firm (The Nordic Council 1992; FTU/VITO/IÖW 1995; Smith *et al.* 1998; Frankl and Rubik 1999). This distinction refers to whether the results of the LCA are just used internally in the firm or are communicated in some form to external subjects. Frankl and Rubik further

* Thanks are due to Frieder Rubik for his substantial contributions.

Table 43.1. Possible uses of LCA in companies

Category	Internal uses	External uses
Research, development & design	Bottleneck identification Comparing existing products with planned alternatives Product innovation and development	
Procurement	Procurement specifications, supplier screening, supply chain management Product stewardship and co-makership	
Production	Process optimization Cost allocation	
Marketing	Comparing existing company products with products of competitors Comparing company product performances with standards and eco-labeling criteria	Marketing, advertising Joining eco-labeling criteria
Information and education	Education and training of employees	Communication, information and education for customers and stakeholders Communication with authorities
Strategy	Strategic portfolio optimization Radical changes in product life cycle Shifting from product to service, creation of new markets Anticipating legislation	Negotiating long-term legislation

suggest that uses of LCA can be classified along the product development chain and can either be *retrospective*, for example, giving just a picture of the existing situation, or *prospective*, leading to innovation and systematic use (Frankl and Rubik 1999, p.32). The latter distinction is discussed below in more detail within the context of the dynamic aspects of LCA introduction and integration patterns in companies. Table 43.1 summarizes the main possible uses of LCA in companies.

Internal Uses

A recent survey conducted on almost 400 companies in Germany, Italy, Sweden and Switzerland shows that, so far, LCA is mostly used for internal purposes (Frankl and

Rubik 1999, p. 72). This conclusion is substantially confirmed by other studies carried out in Nordic European countries (Hanssen 1999) and in Denmark (Broberg and Christensen 1999) and by discussions with other LCA practitioners.

The identification of potential bottlenecks, that is, of environmental critical points along the product life cycle, is currently by far the most common application of LCA in industry. This is not surprising since this is the first step in many kinds of analysis, either retrospective or prospective.

The second most important internal application is the comparison of existing products with planned or possible alternatives (Frankl and Rubik 1999). This does not imply that the results have a strong impact on product innovation. However, in many companies LCA is already used for research, design and development, in particular in Nordic countries. Hanssen notes that 32 per cent of the almost 350 LCA studies reported in recent years in these countries were focused on product development and improvement. It is clear that Nordic countries are currently among the most proactive countries with respect to a systematic use of LCA for environmental product innovation and improvement.

In principle, LCA is also suitable to support long-term strategic decisions. Indeed, it is very desirable that environmental assessment tools be used as early as possible in the product development process, as this can significantly reduce costs. On the other hand, however, 'for more complicated products the number of alternative possibilities is very high, and as the database on exotic materials is limited, the application of quantitative and detailed LCAs to such products may prove to be very resource demanding and at the same time not very precise' (Jensen *et al.* 1997). As a matter of fact, up to 1998 very few companies had used LCA for radical changes in the product life cycle and/or to shift from products to services (Frankl and Rubik 1999). However, the use of LCA for strategy development is expected to increase significantly in the future, as the knowledge on LCA methodology, the internal know-how and data, and the availability of public data-bases improve. Conceptual or simplified LCAs are also more likely to be used than elaborate, detailed, complete LCAs. According to one study, 13 per cent of LCA studies in Nordic countries (25 per cent in Finland) are intended for purposes of strategy development (Hanssen 1999).

External Uses

In the past, especially in Germany, there have been great expectations concerning the use of LCA as a marketing tool. However, even if this was the first motivation for undertaking an LCA, in many cases companies quickly came to realize that LCA cannot be used for marketing at the current level of LCA methodology and availability and quality of data (Frankl and Rubik 1999, p. 255). A similar result is reported for Denmark by Broberg and Christensen (1999). There were several cases in the past in which competitors comparing the same kind of products, using different assumptions, came to opposite conclusions. Of course, in the eyes of consumers, ambiguous or disputed results have low credibility and threaten the credibility of the whole methodology.

Basically, the main problem with LCA is credibility. This in turn is partly a function of the complexity of the results. In many cases in the past there has been a lack of transparency with regard to sources and quality of data, the many assumptions needed (system boundaries, allocation rules, energy mix and so on), the impact assessment method used, subjective valuation factors and procedures, and/or simplification procedures. By changing assumptions respecting these factors, the result of an LCA study can be changed radically. The most crucial point has usually been the availability and quality of data. Unfortunately, in particular in the early 1990s, several of the better known LCAs were shown to be guilty of using 'impossible' data; that is, data inconsistent with mass balance requirements (Ayres 1995a). As clearly stated by Jensen *et al.,* 'This problem, coupled with biased information and lack of quality control, can do more than anything else to undermine the authority of LCA methods' (Jensen *et al.* 1997). The second problem concerns complexity: how to summarize in a simple manner and very concise form a large and complex body of data, information and assumptions. This issue arises even if the execution and the results of an LCA are as transparent as possible. These two sets of problems have limited the use of LCA for external purposes and raised questions about the credibility of the methodology in general.

It is important to observe, however, that there have been very significant improvements in this specific respect in recent years. The ISO norms 14040 and 14041 now set very precise rules for the external communication of LCA results, in particular in the case of product comparisons (ISO 1996, 1997b). Companies are exploring several possibilities for communicating LCA results and obtaining a market benefit in some form. Research institutions and policy makers are participating in this process as well. In this context a particularly interesting initiative is the so-called 'Environmental Product Declaration' (EPD) and its use within the policy framework of Integrated Product Policy (IPP) currently being developed in the European Union. Very clearly, external uses of LCA are expected to increase significantly in the future.

LCA is already widely used to inform and influence suppliers, industrial clients and other stakeholders. The previously mentioned survey carried out in Germany, Italy, Sweden and Switzerland shows that its use to provide customers and stakeholders with generic information and education is the second most important application of LCA (Frankl and Rubik 1999, p. 72). This growing interest in external communication is also confirmed by Hanssen (1999) for the Nordic region and by Broberg and Christensen (1999) for Denmark.

LCA ADOPTION PATTERNS WITHIN A FIRM

Different Analysis Approaches

Several studies have been focused on the integration of LCA in corporate decision-making processes in recent years. Some authors link important aspects of LCA integration in decision-making processes (orientation of life cycle activities, LCA analysts and structure, adoption process and so on) to the specific position of the company within the product chain, that is, whether it produces commodity products, intermediate and simple products or complex products (Berkhout 1996; Smith *et al* 1998). For instance, according to these authors, commodity producers mainly use LCA for external purposes (for example, marketing, policy process), do not use evaluation methods and have a 'top-down' adoption process. At the other extreme, the manufacturers of complex products tend to use LCA for internal decision support, use some evaluation method and have a 'bottom-up' approach. However, this analysis is quite limited, since it gives a static picture of the adoption of LCA in different sectors but at a given time.

In constrast, others point out that the adoption patterns of LCA in decision-making processes are a dynamic sequence of events which imply a learning process and cycles causing significant organizational challenges and changes within the firm (Baumann 1998; Frankl and Rubik 1999). This implies that the role of LCA, the management tools and organizational structures adopted do vary with time and cannot be simply analyzed in a static way. Furthermore, they stress the role and importance of subjective factors in this kind of evolution. See also FTU/VITO/IÖW (1995), Hanssen (1999).

From Learning to Doing: the Institutionalization Theory

Institutionalization theory describes the characteristics of the different phases of the introduction of a new phenomenon (a new idea, a new instrument, an innovation in general) into business activities until it becomes something taken for granted and a routine use. For more details, refer to Tolbert and Zucker (1996). The theory envisages three stages of the institutionalization process. The first stage of application of an innovation within the company is called the habitualization stage. Often it concerns a small part or a restricted area of the company (most likely the environmental department in the case of LCA). The next stage, the one during which the new idea or tool begins to spread out within the company, is called the objectification or semi-institutionalization stage. This is very likely the most crucial phase of the whole process. It is usually at this stage that the future adoption of the innovative idea or tool is determined. If the innovation is further systematically integrated within business activities, one enters the final stage of the full institutionalization process, called the sedimentation stage in the original theory.

The theoretical framework of institutionalization has been used to interpret the results of 20 case studies carried out in Germany, Italy, Sweden and Switzerland in several industry sectors (Frankl and Rubik 1999). More recently, the same theory has been applied to 16 companies in the energy sector in France, Germany, Italy, the Netherlands, Sweden and the UK (Frankl 2000). Figure 43.1 summarizes the results of the case studies by showing the position of the studied companies along the institutionalization–adoption curve. The latter represents the level of adoption of LCA within a company in function of time.

In principle, four possible trajectories of adoption pattern are possible. The upper continuous line in the figure shows an adoption curve leading to the full integration of the innovation (LCA in this case) within the company. The intermediate line represents the case of 'uncertain companies', in which the 'destiny' of the innovation is still unclear. In these companies there are both positive indications which suggest a further integration of the innovation, as well as some negative signals which might indicate a possible future failure. The two other possible (negative) adoption paths – dotted lines in Figure 43.1 – are leading to an early failure or to the late fading out of the innovation. Clearly, the crucial phase is the one of semi-institutionalization, during which the 'destiny' of the innovation is most likely determined.

The first noteworthy result is that, by 1998, just five companies of the 36 surveyed had fully integrated LCA into their decision-making processes. This basically confirms the

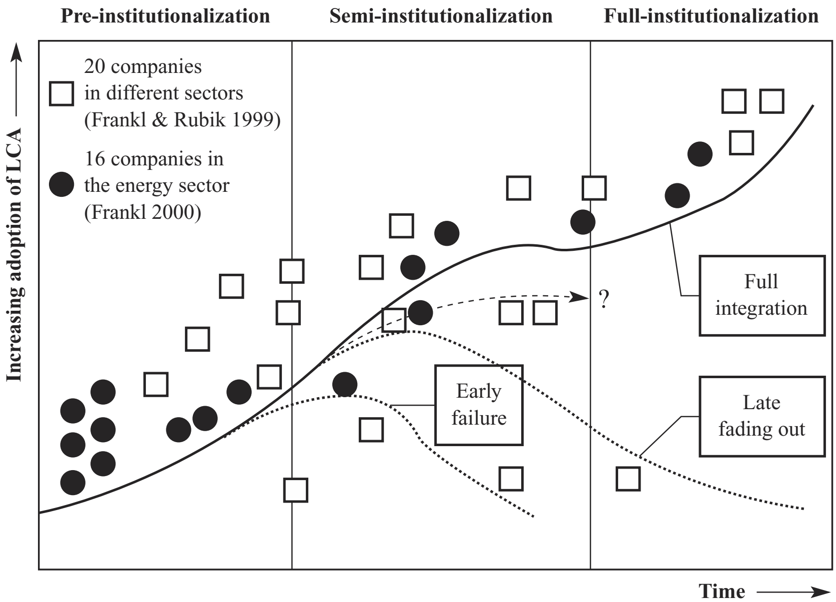


Figure 43.1 Possible adoption patterns of LCA according to institutionalization theory; positioning of 36 surveyed companies by 1998

results reported by other authors, that LCA is still a young methodology and that 'companies have simply not, by and large, felt the need for LCA in their regular decision-making' (Jensen *et al.* 1997, p.14). However, the dynamic analysis approach of case studies and institutionalization theory also clearly indicates that this situation is rapidly changing. As a matter of fact, the eight companies in the semi-institutionalization phase are clearly moving towards full institutionalization in the near future. Moreover, the large majority of firms still in the introduction phase also declared that they will increase the level of LCA activities. In another survey recently carried out in Denmark, 18 companies out of 26 declared that they would base their future product development on LCA studies (Broberg and Christensen 1999).

Success Factors for Full Institutionalization and the Role of LCA

A dynamic analysis approach also allows the identification of the main success factors for full institutionalization of LCA or, conversely, the factors suggesting failure. There are several main factors influencing the institutionalization of LCA in business decision-making processes (Frankl and Rubik 1999). Among the most important is the presence and influence of a 'champion' (or 'entrepreneur'), who pushes LCA activities within the company. Other key factors are the mandate of top management, the involvement of practitioners and development of formalized structures, the establishment of internal communication channels, the development of internal know-how and a long-term environmental commitment.

No particular success factors are needed in the first, pre-institutionalization, stage. As a matter of fact it has been observed that the introduction of LCA can occur either because of a 'top-down' approach (that is, in response to top management directives) or because of a 'bottom-up' approach (for example, by the initiative of an environmental manager, without any mandate from top management at this stage). Which it is does not influence the future adoption of LCA. However the mandate of top management becomes an important factor within the second, semi-institutionalization, stage. At that point, the top management has to be informed about LCA activities and has to encourage their further development.

The motivation and involvement of practitioners is important in all three stages. It might be important as early as the first stage of pre-institutionalization if the introduction of LCA happens 'bottom-up'. It certainly becomes important in the next two stages, as the number of functions and people involved in LCA increases (that is, when other technical departments are involved).

However, the most important single factor in the success of the whole institutionalization process is the presence and influence of a personally committed 'entrepreneur' or 'champion', who pushes LCA activities within the firm. His role is particularly crucial in the semi-institutionalization stage: it is this person who elaborates the strategy to demonstrate the importance of LCA and to create a constituency for it (in particular to obtain the mandate of top management). It is still this individual who has the task of maintaining the mandate and enlarging the consensus by promoting advocacy groups and motivating/involving a larger set of champions during the last stage of full institutionalization. Advocacy groups may not be involved directly in LCA activities, but they support such activities from outside.

Another crucial factor for broadening the consensus and the involvement of people in LCA, thus leading to full institutionalization, is the existence of appropriate (usually inter-department) communication channels. Without such channels, the message of a single champion is inadequate and interdepartmental support cannot be created. This is not important at first, but it becomes increasingly important in the second stage and vital in the third stage (full institutionalization). These communication channels can be created either at the beginning (for example, with a coordination committee for LCA including representatives of different departments) or later on. The latter option is exemplified at Fiat Auto with the 'Integrated Development Plan for Car Components'.

The development of internal know-how (in terms of dedicated human resources, database and software tools) in methods and applications seems to be necessary to achieve full institutionalization of LCA. No company relying only on external support has continued LCA activities in the long term. Of course, the importance of internal know-how increases with time and the institutionalization stage. Formal structures (that is, with full-time dedicated functions) are always present in companies that have reached the full institutionalization stage. Similar observations have been made in Denmark (Broberg and Christensen 1999).

Finally, a long-term environmental commitment seems to be a necessary (although not sufficient) condition for the full adoption and integration of LCA. It was observed that in all successful cases of LCA integration a clear long-term environmental policy was also present in the company.

Quite obviously, the role of LCA changes along the institutionalization trajectory. At the beginning, it is always just for learning. The first application of LCA is typically somewhat retrospective; that is, to confirm already known results. In the semi-institutionalization stage LCA still has a high learning value, but there is a shift from retrospective application towards more prospective uses: LCA is used for designing new products or new alternatives. In the full institutionalization stage LCA becomes a quasi-routine tool. The learning value is consequently much lower. Uses can be very different (internal, external), but tend to be specific. The application tends to be prospective: LCA is used for design choices. In the best cases it is used for everyday decision-making support.

As far as the translation into the practice of LCA results is concerned, the application of LCA influences both short-term and long-term product innovations. This is related to the way learning is translated into action. Usually, short-term innovations are determined by economic factors; changes indicated by LCA are adopted only if they provide economic gains as well. Of course, short-term innovations are important in particular in the full institutionalization stage. However, companies are much more open to the use of LCA in conjunction with long-term product innovations. For instance, LCA results can influence strategic decisions (such as the next cycle of investments) from the very beginning of the institutionalization process.

At present, the most certain and likely highest value of LCA is learning. In all the companies where a learning process occurred there were other changes: a change and development in internal organization, several spin-off initiatives (energy and material saving, bottleneck identification, supply-chain management and so on) and a collaborative attitude tending to compare experiences with other companies and/or research institutes. Many firms adopted the goal of using LCA in a systematic way in the future as a support tool for decision making, eventually using simplified methods or procedures (Frankl and Rubik 1999). The value and importance of learning are also highlighted by Broberg and Christensen (1999), who indicate that in many cases the work on LCA actually results in new priorities for companies.

FUTURE OUTLOOK AND DIFFUSION PROSPECTS

General Observations

The general outlook concerning the future use of LCA as a management tool is very positive. Already today, LCA is diffused in most industry sectors in Europe, the USA and Japan, other Asian countries, and also in Australia. European countries and Japan are clearly at the forefront of the methodology and its application in industry. As far as we can determine, currently Germany has the largest absolute number of LCA studies, but Sweden has the highest in relation to GDP.[1] LCA activities are increasing very rapidly and significantly in Japan, where there is an important National LCA project (Hunkuler *et al.* 1999; Finkbeiner and Matsuno 2000). The large majority of surveyed companies declare that their LCA use will increase in the future, possibly in combination with other tools (Frankl and Rubik 1999, p. 82). There is a clear shift from a rather retrospective use towards a more prospective use leading to product innovation and systematic use as a decision-making support tool. Moreover, an increased use for external communication is expected (see below).

Open Issues and Possible Solutions

The most important unresolved issue for LCA today is still the availability and quality of data. In particular, this may be an insurmountable barrier for small- and medium-sized enterprises (SMEs), which have neither financial nor human resources to obtain or develop these data. As far as quality is concerned, crucial points are usually the lack of transparency, mismanagement of missing data, insufficient disaggregation of data and inflexibility and/or secrecy of the data-base (and/or the LCA software tool). A very significant contribution to the solution of this problem can be made by the development of a public shared data-base at the national and international level on some specific parts of the life cycle which are common to practically all products. Examples might include energy systems, transport, waste management and production of bulk materials such as plastics or steel. This is exactly what is happening in several countries in Europe, Japan and Korea (Frankl 2000; Hunkuler *et al.* 1999). A second very important step forward is systematic verification by external peer reviews, as indicated by the ISO norms. Of course, the latter has to be done at a high level of excellence.[2] In any case, transparency and credibility can be significantly enhanced by involving stakeholders in external/public LCA studies as early as possible (Wrisberg *et al.* 1997).

A second unresolved issue is related to impact assessment and indicators. Very significant progress has been achieved recently in this area, in particular with respect to the assessment of local impacts (see Chapter 12). Nevertheless, the problem still remains that the weighting of single impact categories is, and will always be, a subjective procedure, depending on local and human factors. In this respect, it is worth mentioning the possibility of using other tools, such as multi-criteria analysis, for the interpretation of LCA results (as, for example, in Hirschberg and Dones 1998 for energy systems). This kind of model and approach is an extremely important example of the way LCA can be used for decision making. It combines different indicators in order to arrive at a final evaluation, but without reducing everything to a single unit of measurement (money or 'points'). Instead, it reflects the complexity of the real world and takes into account the importance of subjective valuations. The assumptions are completely transparent; the system is fully flexible and necessary sensitivity analyses can be performed. The other advantage is that it can include not only environmental aspects but also health risk assessment, and social and economic factors in the final evaluation (see below).

Asking the Right Question: the Link with Other Management Tools

The question is not whether LCA will become the ultimate environmental assessment method of products and services taking all possible effects and impacts into account. On the contrary, the question is: under what conditions will LCA be a valuable support tool for management? This is the real question for the decision makers, at any level, either in business or in public administration.

The answer to the first question is clearly 'yes'. Despite all the complexity and the problems still remaining, LCA is already a very valuable support tool, given that some key requirements (maximum transparency, flexibility and openness of the model) are satisfied. However, it should be very clear that LCA cannot and will never be able to replace the decision maker himself. LCA simply quantifies the complexity of reality. The value of

LCA as a support tool for decision making is that it gives a strategic system view while not glossing over the technological details, it has an enormous learning value and it can reduce the number of parameters for which decisions have to be taken.

The link with other (environmental and non-environmental) management tools is of paramount importance. So far, there has not been a clear correspondence between LCA and other environmental management systems (EMS) such as ISO 14000 and/or EMAS. The existence and/or planning of an EMS seems a necessary, but not sufficient, condition to carry out an LCA (Frankl and Rubik 1999, p.59). In certain countries, where EMS were strongly focusing on the organization or the site only, the relation between LCA and other EMS is particularly weak (Broberg and Christensen 1999). However, the positive relation with LCA is expected to increase strongly in the future; for instance, EMAS II regulation explicitly focuses on the life cycle of products and services.

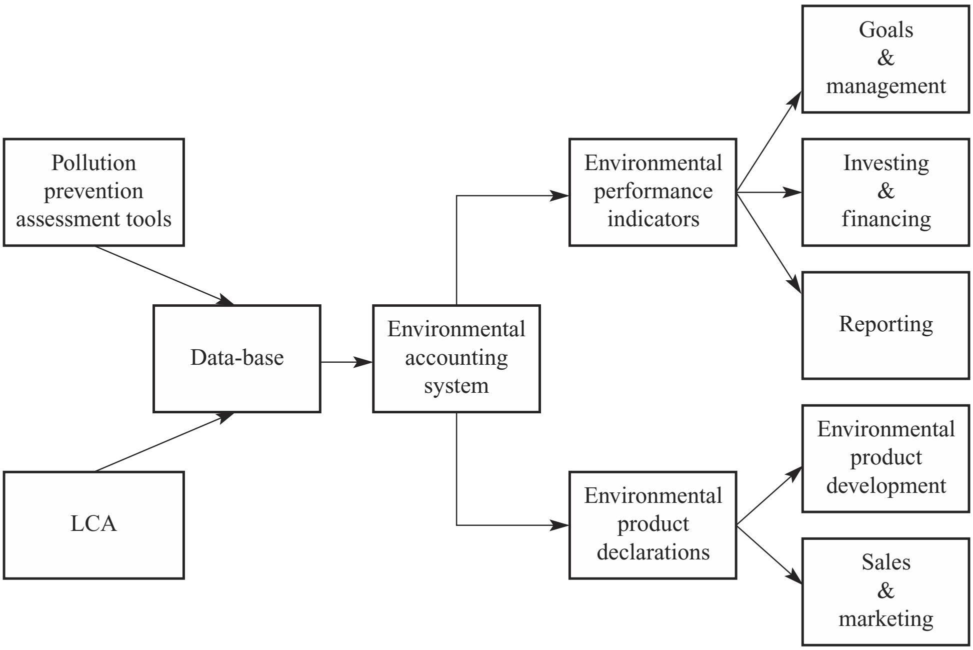
From a Product to a System Approach: the Role of LCA in Industrial Ecology

So far, LCA has been mostly used for the environmental assessment of products. In recent years, it also has been used to assess some services. This is a first step towards the applicability of LCA for eco-efficiency. However, the real potential is the network approach to environmental problems typical of industrial ecology. This requires a paradigm shift from a product approach towards a system approach. Moreover, it implies a shift towards the assessment of sustainability, including social and economic factors, and not just the environmental impacts. This means that innovative assessment and communication methods and tools have to be developed in order to address different stakeholders at the same time. In particular, these tools have to be able to reflect the value of sustainability in market terms (for example to financial stakeholders and/or consumers). This process has barely begun, but it represents a most important potential of LCA as a management tool.

The Importance of Effective Communication

In such a network approach effective, different and specific assessment and communication instruments are needed. Different tools have to be used for different target stakeholders. The model proposed by Hanssen (1999) is relevant. In his view, life cycle data have to be managed in specific environmental accounting systems, in a manner similar to that in which economic data are treated today. 'From these databases, the companies can communicate information internally and externally through Environmental Performance Indicators with focus on organizations, and through Environmental Product Declarations with focus on specific products' (Hanssen 1999) (see Figure 43.2). As already mentioned, the latter, based on life cycle, third party-verified quantified information is rapidly gaining interest in several countries. In 1999, the Swedish Environmental Management Council created an EPD logo and fixed the criteria for companies to obtain the label. This is expected to happen also in Italy at the beginning of 2001. EPD is currently the object of a standardization process (ISO-TR 14025).

The proposed model involves different stakeholders, with different tools for communication. Of course, several other models are currently being proposed as well. For instance, in the EPD of its cars, Volvo combines information about the product and the organization,



Source: Hanssen (1999).

Figure 43.2 Possible life cycle-based management toolkit and communication flows

notably on environmental management (Volvo Car 2000). All the proposed models follow a multi-stakeholder approach.

A Multi-stakeholder Approach: Success Factor for the Future Diffusion of LCA

A multi-stakeholder approach seems to be necessary to implement the shift from a product approach to the systems approach appropriate to industrial ecology. In the particular case of LCA, multi-stakeholder approaches might play a crucial role as a success factor for the future diffusion of LCA in any case. There are several reasons for this. As mentioned above, LCA still has a set of important open issues, that is, availability and quality of data, methods of impact assessment, simplification and screening procedures, and so on. Even large companies feel the need to exchange and compare their experience with other companies and research institutions. If there is a place where this interaction can happen, this turns out to be a very important factor for the success of LCA. In fact, there is an example of this kind in Sweden, the Centre for Environmental Assessment of Product and Material Systems (CPM) at Chalmers Institute of Technology in Göteborg. This center has been collecting all the experiences on LCA from different sources (companies' research centers, universities and so on. It is a reference clearing-house for all LCA practitioners in Sweden, and has played and plays a crucial role in the diffusion of LCA in that country. Similarly, it is worth mentioning that national associations of LCA have been founded in recent years in Japan, Korea (for details, see Hur 1999), India and Italy, which include representatives from industry, universities and research institutions, NGOs, consumer organizations and public authorities. Also, in France, there is an initiative to create a French association for LCA, based on the positive experience of RECORD (focusing only on waste management).

In summary, it is likely that multi-stakeholder organizations will play a crucial role in the future diffusion of LCA with respect to the following:

- availability and quality of data (either peer reviewing or commissioning shared data-bases;
- impact assessment methods (defining guidelines with respect to indicators to be used and methods to evaluate the different impacts – for example, multi-criteria analysis);
- developing standardized simplification and screening procedures;
- facilitating the dialogue between business and legislators, thus enabling business to better anticipate legislation;
- developing methods to combine LCA with other management tools for decision-making support;
- exploring ways of using LCA for environmental declarations of products, third party-certified eco-labeling and/or environmental product information schemes in general, within the policy framework of IPP;
- (last but not least) fostering the education of consumers, who will play an increasingly important (and potentially crucial) role with respect to more sustainable consumption and industrial ecology.

Creating a permanent dialogue between all the involved stakeholders (in particular, between suppliers, regulators and consumers) and sharing responsibility among them is of crucial importance for the future diffusion of LCA and its growing integration within the framework of industrial ecology.

NOTES

1. Most LCA studies are not published. A systematic updated inventory of LCA studies does not exist. Frankl and Rubik (1999) refer to almost 300 studies by the end of 1997 in Germany. However, another author estimates that by May 2000 the number of LCA studies was well over 500 (K. Saur, president, SETAC Europe, private communication, May 2000). Hanssen (1999) reports almost 350 LCA studies carried out in Nordic countries alone.
2. SETAC-Europe organized a case study workshop called 'Increasing Credibility of LCA' in November 2000, where many of the problems and difficulties related to peer reviews and communication of LCA results were discussed (Weidema 2000).

44. Municipal solid waste management

Clinton J. Andrews

People generate garbage as they live their lives, as they produce and consume. In the process of disposing of this garbage they in turn generate many headlines, signs of unresolved controversies. Waste disposal has been a concern for as long as there have been human settlements, and current debates have ancient origins. This chapter examines municipal solid waste (MSW) management from the industrial ecology and political economy perspectives. It excludes industrial wastes (see Chapter 32) and construction-related wastes, and focuses primarily on household and small commercial waste streams. Since the field of industrial ecology is motivated in part by dissatisfaction with current waste management practices, this chapter also considers implementation issues affecting industrial ecology.

Solid waste management involves both public and private actors, and cultural, political and economic judgments. Over historical time the definition of trash has been a moving target. Current management practices are best understood in historical and politico-economic contexts.

HISTORICAL BACKGROUND

Archaeologists have identified persistent themes in MSW management. For example, Bronze Age Trojans periodically became so offended by household waste that they covered it with clay; ancient Mesopotamian cities were invariably located upwind of remote garbage dumps; in Old Testament Jerusalem, people incinerated their garbage in the nearby valley of Gehenna (later a synonym for 'hell'); and the wealthy Classic Maya generated more reusable and recyclable trash than their poor Late Post-Classic descendants (Rathje and Murphy 1992).

For most of recorded history, household wastes not left on the floor were simply thrown outside the dwelling to decompose or to be scavenged. In rural areas this was not troubling because little other than food waste and ashes was thrown away. Used clothing, furniture, tools and weapons were carefully repaired or handed down, even in wealthy households. Rags were recycled for paper making. Some organic wastes such as fat and bones were used by households to make soap, candles and other items. In cities, similar patterns prevailed, with the addition of loosely organized groups of scavengers who recovered items of value from the street (Strasser 1999).

By the mid-18th century, rural Americans began digging refuse pits, and urbanization in cities like London and Philadelphia led to crowded conditions that made ad hoc waste disposal untenable, so public street cleaning began (Rathje and Murphy 1992). It took another century and a spate of cholera epidemics to make sanitation a fully accepted public responsibility in cities such as Chicago, New York, London and Hamburg (Gandy 1994).

The 19th century also brought large-scale industrialization and *laissez-faire* capitalism to Europe and North America. With them came mass-produced products, widespread participation in the cash economy by women, less household self-sufficiency, a disruption of historical patterns of recycling, repair and re-use, and an absolute avalanche of municipal solid waste (Strasser 1999). Rathje and Murphy (1992, p. 41) quote the historian Martin Melosi: 'one of the great ironies of unbridled laissez-faire capitalism was that it gave rise to a kind of "municipal socialism" as cities were forced to shoulder responsibility for such duties as public safety and sanitation'.

Scavengers have not disappeared, so that a tension persists to this day between the public and private aspects of waste management. Waste could be privately managed, but it is often publicly managed to reduce the negative externalities. Even if it remains a public responsibility, either public employees or private contractors could perform the work. We could dispose of waste in the most convenient, least costly way possible, or we could thoroughly mine it for valuable or hazardous constituents. New actors motivated by outrage at wasteful consumerism, and by concern for environmental sustainability, have added a moral dimension to the debate. Rapid technological change is further complicating matters, as explained below. Many parts of the developing world are still making the transition to a formal, institutionalized scheme for solid waste management (Onibokun 1999).

Dumping, incineration, resource recovery and source reduction have been the key waste management strategies throughout history, and they remain with us today. Yet dumping has evolved rapidly over two centuries from tossing household waste out of the front door, to placing it in a hole in the back yard, to delivering it to an open and unlined dump, to burying it in an engineered, lined landfill with methane collection. Incineration has evolved equally rapidly during the same period from open burning in the back yard, to large-scale urban 'cremators' that controlled combustion very poorly and generated noxious smoke, to slightly cleaner second-generation incinerators, to modern incinerators with emissions controls and energy recovery. Thermochemical reduction technology is a century-old variant that recovers useful products; there are also biologically based processes ranging from traditional composting to designer microbes. Recycling, remanufacturing, repair and re-use are ever-present activities that used to take place primarily within the household and now involve centralized sorting facilities and extensive networks of economic and political actors (see Andrews and Maurer 2001 and Chapter 41). Source reduction, formerly the unwilling choice of the newly poor, has evolved into a management strategy that combines behavior modification techniques and engineering design tools. The opportunity to choose among these strategies ensures controversy because their relative strengths and weaknesses change rapidly.

The waste stream itself also changes rapidly, as new materials, products and services are invented, and as people's wealth and aspirations grow. Over the past century in the USA, for example, its magnitude has multiplied and its primary components have shifted from food waste and ashes to yard waste and paper. The changes continue, as is discussed below.

POLITICAL ECONOMY PERSPECTIVE

Rapid changes in both the character of the waste stream and in the relative attractiveness of management strategies demonstrate that 'trash is a dynamic category' (Strasser 1999, p. 3). Since trash is created by sorting, nothing is inherently trash, and it becomes important to focus on the categorizing process that defines trash (ibid., p.5). Solid waste management involves both collective decisions and individual decisions nested within the collective framework.

Societies make collective decisions about solid waste in three overlapping areas. Cultural collectivities decide religious, moral, linguistic and aesthetic matters, and they establish boundaries on political and economic decision making. Politics is about the way human beings in aggregate are governed, and this involves ordered structures, the exercise of power and claims regarding justice (Magstadt and Schotten 1988). Economics is about the way in which resources are allocated among alternative uses to satisfy human wants (Mansfield 1970), in a world with working but imperfect markets.

Political economy, as used here, examines 'the interrelationship between the practical aspects of political action and the pure theory of economics' (Pearce 1986). The strand of political economy research that examines the provision of public goods is especially helpful for thinking about solid waste. It analyzes collective decision making using a microeconomic lens, and its vocabulary includes terms such as excludability, divisibility, externalities, asset specificity and transaction costs (see Chapter 5 for definitions). The issues are illustrated primarily by using US data, in order to keep this chapter tractable; the details of other systems differ and are well described elsewhere (for example, Onibokun 1999; Gandy 1994; Carra and Cossu 1990; Barton *et al.* 1985; Bridgwater and Lidgren 1981).

CURRENT MUNICIPAL SOLID WASTE MANAGEMENT PRACTICES

Today, municipal solid waste (MSW) is a stream that comes primarily from residences and small commercial enterprises, and is mostly non-hazardous. According to the US Environmental Protection Agency definition followed here, MSW includes durable and non-durable goods, containers and packaging, food wastes, yard wastes and miscellaneous inorganic wastes from residential, commercial, institutional and industrial sources (USEPA 2000). The EPA definition excludes industrial waste, agricultural waste, sewage sludge and hazardous wastes including batteries and medical waste. Americans generate increasing quantities of municipal solid waste, whether measured in total tons or kilograms per capita (USEPA 2000). Containers and packaging are the largest fraction of this waste stream (33 per cent in 1995) and non-durable goods are the most rapidly growing fraction (USEPA 2000).

Disposing of municipal solid waste has become increasingly costly and centralized. In the early 1970s, municipal solid waste disposal was primarily a local matter, and most was sent to one of more than 18000 town dumps (*Waste Age* 1974). Governmental oversight of environmental, health and safety issues related to landfills was minimal, and highly visible problems developed. In 1976, with important amendments in 1984, the US

Congress passed legislation that led to greatly tightened restrictions on the environmental performance of landfills. Tighter regulations were slowly phased in over several years. By 1987, the total number of landfills had decreased to 6034, with 1122 large landfills (over 175 tons per day) capturing the vast majority of MSW (USEPA 1988). By 1998, there were only 2314 landfills, yet because many are large the nation's total landfill capacity has remained roughly constant (USEPA 2000).

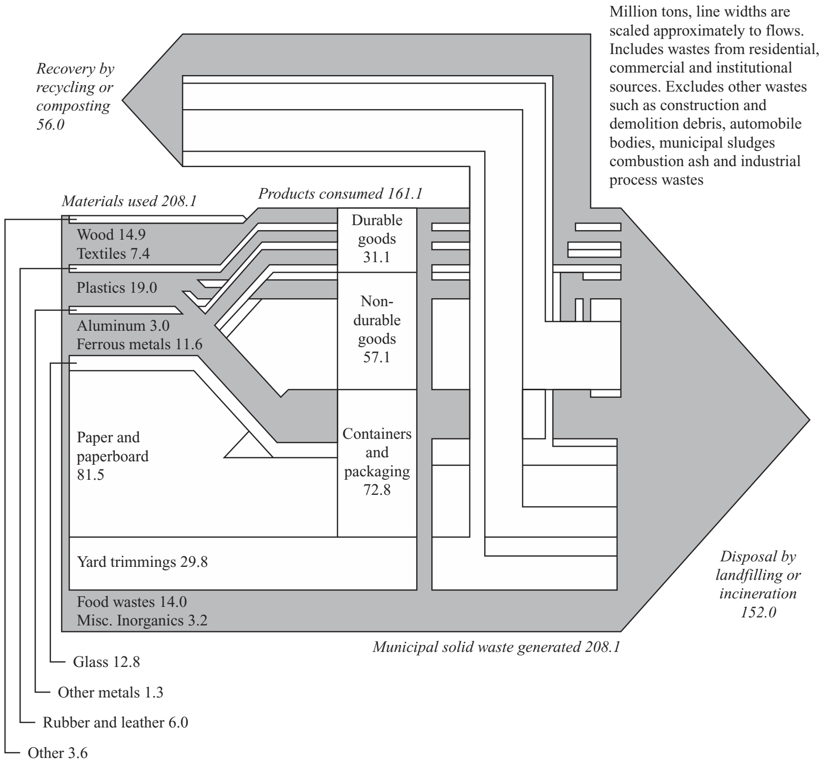
High landfilling costs and stricter regulations have spurred various technical and behavioral innovations in municipal solid waste management. Technical innovations include incinerators with waste-to-energy capabilities, materials recovery facilities to recover high-value items, and composting. Behavioral innovations include recycling programs and a variety of strategies aimed at source reduction, such as education and pay-as-you-throw disposal fees. Today the vast majority of MSW is still landfilled, but some is burned and an increasing fraction is recovered through recycling or composting (USEPA 2000). Much diversion by means of re-use, repair, remanufacturing and recycling takes place for economic and cultural reasons, but public policies further encourage these efforts, as other chapters of this handbook attest.

Municipal solid waste policy makers have a long history of performing industrial ecology-style analysis, examining impacts of disposable products and the role of producers during the 1950s, mass flows starting in the 1970s and life cycle assessment in the 1980s (Gandy 1994). Thus the analytical tools that industrial ecologists use have already been accepted in this policy community, making it a good case for examining political and economic issues affecting implementation.

Figure 44.1 shows the amounts and fates of the various materials comprising the US municipal solid waste stream in 1995. Several features stand out. First, only about one-quarter of the waste stream is recovered through either recycling or composting, suggesting great potential for future loop-closing activities. Paper and paperboard is the largest fraction of the waste stream by weight (39 per cent of MSW), and a relatively large fraction is recovered (40 per cent), suggesting that much effort has already been applied to this most obvious target. High-value metals enjoy similarly large recovery efforts (39 per cent in aggregate is recovered), while 30 per cent of yard trimmings are recovered (mostly by composting). Glass recovery is less successful, at 25 per cent, and plastics lag even further behind, at 5 per cent. Food wastes stand out as the material with the lowest recovery rate (4 per cent). The different recovery rates for various materials suggest that each has a unique set of technical, economic and policy drivers.

A product life cycle view of MSW is shown schematically on the left side of Table 44.1. Analysts also bemoan the problem of having responsibility for product management divided by life cycle stage (Schall and Wirke 1990; Graedel 1994) and they have endorsed solutions including packaging and product take-back (Lindqvist and Lifset 1998) and a product leasing business strategy (Stahel 1994).

The industrial ecological tools of mass flow analysis and life cycle analysis yield general insights – close more loops, do source reduction, design for the life cycle – yet they do not illuminate all current policy controversies. Headlines are instead about transboundary transport of waste, deregulation of a heavily regulated industry, and environmental justice, among other items. Many of these factors impede implementation of the industrial ecology vision, but that vision seems blind to the forces that generate the headlines. Political economy analysis clarifies the implementation issues.



Source: Characterization of municipal solid waste in the USA, 1996. Update prepared for the US Environmental Protection Agency by Franklin Associates, Prairie Village, KS, June 1997.

Figure 44.1 US municipal solid waste flows, 1995

ROLE OF POLITICAL STRUCTURE

The structure of a nation's political system strongly affects the way its solid wastes are managed. Affected are the scale of operations, patterns of materials flows, financing arrangements, standards of performance and the pace of regulatory reforms. A brief review of each level of government's responsibilities follows for the US case. This discussion of the political economy of solid waste is based on Chertow (1998), USOTA (1989), Rabe (1994), Williams and Matheny (1995) and Luton (1996).

Many local governments have provided residential MSW collection and disposal as a public service funded out of general tax revenues. They also require by statute that commercial enterprises dispose of waste in an acceptable manner. Local governments choose from a spectrum of provision options that includes public provision, public contracts with private providers, granting of monopoly franchises, regulated private competition and no public involvement. There has been a fairly clear trend over recent decades towards privatization. Some municipalities emphasize recycling, while others do not. Local governments also typically set land use regulations and site facilities such as incinerators.

Table 44.1 Actor by life cycle stage

Actor	Producer (Manufacturer)	Consumer (Household)	Waste Management Industry	Advocacy Organization	Local Government	State/ Provincial Government	National Government
Life-cycle Stage							
Product Design	X					X	X
Raw Materials	X						X
Manufacturing	X						X
Packaging	X						X
Distribution	X						X
Marketing	X			X		X	X
Purchase		X		X			
Use		X		X			
User Actions		X	X	X	X	X	
Payment		X	X		X	X	
Collection			X		X	X	
Local Processing			X	X	X	X	
Regional Processing			X			X	
Disposal			X			X	X

Note: X marks life cycle stages where actor has greater influence.

Some states have chosen not to delegate all of these responsibilities to localities, giving counties a significant role. States regulate waste haulers and disposal operations. States also require counties and/or municipalities to develop solid waste management plans, and some go as far as requiring at-source separation and recycling, in addition to ensuring adequate incinerator or landfill capacity. States have also passed bottle bills and other legislation affecting the solid waste stream.

At the federal level, legislation and subsequent regulations have set stringent standards for landfill design, incinerator emissions and the exchange, transport and storage of solid wastes, especially those classified as hazardous. Recently, the US Supreme Court has invoked the inter-state commerce clause of the Constitution to invalidate state-level solid waste management flow control laws that directed in-state waste flows to specific in-state facilities (Gordon and Weintraub 1995).

In short, all levels of government are involved in MSW management, with higher levels establishing the rules, and lower levels implementing them with a great deal of autonomy. This places MSW management, and the implementation of industrial ecology, firmly in the context of the reigning political structure (US federalism in this case).

A useful, politically oriented advance in the life cycle assessment process is to consider the various potential actors at each life cycle stage (King 1996). The key actors include producer industries, waste management firms, household generators, commercial generators, institutional generators, community organizations/non-profits, municipal government, county government, state government and the federal government. Each is very involved at some point in the product life cycle, but may have little impact on other parts of it. Table 44.1 shows that there is a vast range of solid waste management options by actor and life cycle stage. The figure reminds us that policy needs more than a 'what' and 'why' – reformers must also answer the 'who' question. These questions are interdependent, and industrial ecologists cannot meaningfully answer the first two questions without a deeper understanding of the third.

A particularly contentious feature of units within federal systems is their *interdependency* (Rabe 1997). All else equal, the political actors in these units prefer as much independence as possible, but economic and geographical forces create binding ties. Controversy can erupt when *unbalanced* relationships develop which seem to transform interdependence based on comparative economic advantage into dependence that decreases local political autonomy. Over the last 30 years, the 50 US states have gone from nearly complete self-sufficiency in municipal solid waste disposal to a situation of persistent imbalance (Goldstein 2000). Persistent imbalance is not always seen as a problem: few complain when New Jerseyans buy cars from Michigan and avocados from California. However, local environmentalists are less willing to let the state's municipalities bury waste in Pennsylvania because it puts environmentally significant activities beyond the reach of the New Jersey state regulatory apparatus.

Federal systems of government may promote inefficiency when policies run counter to market forces. Some states have wielded a strong regulatory hand in the municipal solid waste arena, visible in the adoption of flow control policies, for example. A dramatic price gradient persists even as we enter the post flow-control era in the USA (Repa and Blakey 1996). While the higher land and energy prices in the northeast region contribute to higher MSW tipping fees, alone they cannot account for weighted-average regional tip fees that differ by *hundreds* of per cent. Dramatic variations in average tip fees persist even between counties in one small US state (Andrews and Decter 1997). Market failures and/or politics must play a role.

BASIS FOR GOVERNMENTAL INTERVENTION

Political economy offers arguments for determining which role each collective decision-making mechanism should play in implementing the industrial ecology vision. Solid waste management in the USA involves both economic and political actors, but its public service aspects are diminishing and its private industry aspects are increasing. During this transition, there has been minimal consideration of the characteristics that locate it on the continuum from public to private goods. Here the collection, processing and disposal components of the MSW sector are tested briefly against criteria from the theory of public goods. Table 44.2 summarizes the results: a 'Yes' answer indicates that there is a basis for government intervention.

Inexcludability?

No, the entire MSW management process enjoys excludability, since service providers can if necessary withhold access to the system. However, the system boundaries as defined by a political economy analysis differ from those defined by an industrial ecological analysis. Symptomatically, the sector has a history of illegal disposal activities that take place outside the 'system' and impose social costs. Government therefore must intervene as an enforcer of laws (primarily for commercial generators) or as a universal service provider (most often for residential generators).

Table 44.2 Economic characteristics of municipal solid waste industry segments

Characteristic	Definition	Collection	Processing	Disposal
Inexcludable?	Cannot prevent consumption by those who do not pay for product	No. Can prevent. But local gov't may want to ensure universal service	No. Can prevent	No. But need to enforce laws against illegal disposal
Indivisible?	Costs the same for two to consume product as for one	No. Costs more	No. Costs more	Yes. There are significant economies of scale
Externalities?	Spillover effects from transaction that affect others	Yes. Uncollected or sloppily collected waste is offensive to neighbors	Yes. Processing sites generate noise, odors, traffic	Yes. Disposal sites generate noise, odors, traffic, stigma
Asset specificity?	Has specialized use, is not adaptable	No. Vehicles are adaptable	No, except for high-tech MRFs. Transfer stations and simple MRFs are adaptable	Depends. No for landfills and composting; yes for incinerators
Transaction costs?	Completing a transaction requires much time, money, effort or information	Yes. Many parties, many options and much ignorance make transactions costly	No. Few facilities and options make transactions easy, except for immature recycling markets	No. Few facilities and options make transactions easy

Indivisibility?

No, there are positive marginal production costs for all components of the MSW system. However, there are dramatic scale economies in the disposal component which make short-run marginal costs small (USEPA Office of Solid Waste Management 1995). This is in part due to high fixed costs associated with the start-up and shut-down of landfills and incinerators. In addition, landfill capacity increases with the volume of the site, whereas operating and environmental protection costs increase with its surface area. Thus large landfills thrive in an era of more stringent environmental protection, just as large mammals like whales and polar bears thrive in cold climates, because surface area-to-volume ratios decrease as size increases. Support for this relationship is evident in the order-of-magnitude drop in the number of open landfills over 20 years without a corresponding loss of national disposal capacity (ibid.). Modern incinerators equipped with air pollution control and waste-to-energy conversion equipment also enjoy significant scale economies, with the economically optimal size currently in the 1200 tonnes per day range (ibid.). At the present US average MSW generation rate of two kilograms per day per person (USEPA 2000), such an incinerator should serve the needs of 600000 people. That is more than reside in most municipalities and many counties. By contrast, the collection and processing components of the MSW management system appear to enjoy fewer scale economies, although collection enjoys economies of contiguity (see below). A result is conflict between collective decision-making mechanisms: the marketplace would like to organize disposal differently than would local governments. The trend towards regional disposal facilities reflects government's belated interest in the scale economies.

Externalities?

Yes, the sector clearly suffers from pervasive negative externalities (Schall 1992; Schuler 1992). Uncollected waste quickly becomes offensive to neighbors. Illegally disposed waste also offends. Collection vehicles converging on processing and disposal sites increase air pollution, noise and congestion. Processing sites (materials recovery facilities, transfer stations) generate noise and odors. Landfills generate noise, odors, dust, air pollutants including greenhouse gases, groundwater pollution potential, and may stigmatize host communities. Incinerators add potential air pollution problems to this list. As a result, the sector is heavily regulated, and governments often take an ownership or management role. Yet public decision makers realize that some externalities can be exported (such as those associated with landfills and incinerators) while others cannot (such as those associated with collection). Local governments thus may consciously create 'territorial' externalities that can only be internalized with help from higher levels of government.

Asset Specificity?

Yes and no. Modern incinerators rank with nuclear power plants in having extreme asset specificity. These capital-intensive, immobile, highly optimized machines are very good at converting waste to energy, but they have little value for anything else. They are very risky investments in an uncertain world, and governments have stepped in to help investors manage these risks. In 1993, some 58 per cent of US incinerator throughput was tied to captive markets through the mechanism of flow control, and another 31 per cent was guaranteed under long-term contracts, many with put-or-pay clauses, guaranteeing revenues regardless of waste flows (USEPA Office of Solid Waste Management 1995). A majority of the remaining 11 per cent of throughput went to publicly owned facilities. The demise of flow control has left these facilities exposed to market risks, with the result that many have been tagged 'stranded investments', instruments used to finance them have been downgraded to junk bond status and interest in new facilities has waned (Andrews and Decter 1997). Landfills and composting facilities suffer much less from the asset specificity problem because, once the site is purchased, remaining capital investment occurs only as needed over a period of years. In addition, former landfill sites frequently find subsequent uses. Transfer stations and materials recovery facilities typically have low-asset specificity because they are low-cost, generic industrial structures: a shed, concrete floor and truck bay. Collection vehicles likewise have low asset specificity. The policy question that some state and local governments only recently started asking is: are the benefits of large investments with high-asset specificity worth the risks?

Transaction Costs?

Yes, transactions between waste generators and waste collectors carry significant costs. Many municipalities contract with collectors on behalf of their citizens to spare them the trouble of doing so directly, and to capture 'contiguity economies' by avoiding the redundancies of route competition. Dense networks of highways and rail lines have meanwhile reduced waste transport costs to the point where large generators (firms, municipalities) in many areas can choose among several disposal options. Recycling suffers from high transaction costs because of the need to collect and manage many small materials streams, and because secondary materials markets are not well established. Transaction costs between collectors, processors and disposers are relatively low, suggesting that this industry should have a low level of vertical integration. Horizontal integration appears to be the dominant mergers-and-acquisitions strategy of this industry (*WSJ* 1998). Transaction cost concepts help to explain some of the structures seen in this industrial ecological community. A final important role of transaction costs is in the enforcement of solid waste laws; thus the high costs of preventing illegal dumping and anti-competitive business activities make it easier to understand why segments of the MSW industry have been riddled with criminal activity.

In sum, these economic characteristics of the solid waste industry explain why serious market failures should occur, and the historical record confirms that they do so. A pervasive governmental role in the sector seems unavoidable. However, given evidence of improbably steep price gradients, it appears that current political decision-making bodies have sometimes substituted expensive government failures for the market failures. As the constituents of the waste stream change and the technologies for managing waste evolve, it periodically becomes necessary to adjust the relative decision-making roles of politics and the marketplace.

IMPLEMENTATION LESSONS

Several implementation lessons follow from the above analysis. Since the political economy perspective forces analysts to tailor their messages to a specific group of decision makers, the focus here is on guidance just for local officials.

First, the analysis suggests that, as local government officials lose coercive fiscal authority owing to the demise of policies like flow control, they need to become better risk managers. The task has several aspects: learning to recognize asset specificity and similar vulnerabilities, shedding risks by contracting for waste management services, designing flexibility into contracting arrangements and ensuring public buy-in on decisions made on the public's behalf. In many circumstances the industry should raise capital and organize its industry structure with less governmental involvement.

Second, NIMBY (not in my back yard) pressures may help to prevent wastefully duplicative investments in disposal capacity, provided that the legal tools wielded by community opposition groups are not undercut (Luton 1996). Environmental justice is gaining an increased focus even as the sector shifts to greater reliance on economic rather than political decision making. Yet if environmental performance of the sector improves, persistent interjurisdictional MSW trade imbalances should not be troubling, any more than they are for food or motor cars.

Third, the policy goal of loop closing can be disentangled from the job of regulating an increasingly efficient waste collection and disposal industry. There should be no need for the elaborate cross-subsidy schemes often seen in vertically integrated systems that use landfill profits to pay for recycling. Unbundled costs and full accounting transparency can become the basis for a more legitimate contracting strategy. Governments can contract explicitly for curbside recycling as part of a MSW collection package, and encourage additional loop closing through greener procurement policies. While recycled content procurement directives may seem like blunt instruments compared to full life cycle assessment guidance (currently unavailable), they are much more efficient than current strategies that only look downstream and rely on a coercive local recycling process.

The life cycle analogy that connects the production, consumption and disposal of goods and services suggests that municipalities should send specific signals upstream to waste generators, and downstream to waste managers. For example, municipalities could implement 'Pay-As-You-Throw' (PAYT) to send signals upstream to households (Miranda *et al.* 1994). The property tax levy has been the predominant method of financing municipal solid waste services in the USA, and other nations also rely on fixed charges of various types (Gandy 1994). Thus the amounts paid by residential waste generators do not relate to the amount of waste they generate. As a result, there is little economic incentive for waste reduction or for increased recycling participation rates. By pegging user fees' rates to the amount of waste generated, appropriate signals can be sent to affect consumer behavior, and work their way further upstream to affect production and marketing choices (Jenkins 1993). Unit pricing or PAYT programs are not exotic: they have become an integral and successful part of municipal solid waste management systems in more than 4000 US communities (Hui 1999).

Actors can likewise use contracting to send signals downstream to waste managers. Municipalities and their residents contract for waste management with firms or other governmental units. In the US post-flow control era, contracting has to be open and nondiscriminatory on the one hand, while achieving a variety of economic, environmental and institutional goals on the other hand. To ensure political legitimacy, municipal officials can develop contracts that reflect community values. Such contracts could include recycling and education programs as well as waste disposal.

Since ancient times, households and businesses have engaged in economically motivated recycling, remanufacturing, repairs and re-use. Governmental actors should make their waste management activities more complementary to these private actions, and be wary of replacing them unnecessarily.

Policy recommendations for other levels of government would be quite different. Speaking broadly, state and local governments should seek more of an oversight role and less of a provider's role in the MSW industry. If government can adequately police illegal dumpers and remove barriers to regional disposal markets that exploit scale economies, waste management can take on more of the characteristics of private rather than public goods. Then regulated private firms will more often become the logical providers. Economic regulation of the solid waste industry can move to the national level, where anti-trust lawyers prosecute individual cases of excessive market power. Meanwhile, the environmental regulation of this industry would have to remain stringent. State and local governments would need to continue enforcing strict performance standards for waste collection and processing, and actively prevent illegal dumping. The federal government would have to preserve, and state governments would have to enforce, current stringent standards on landfill and incinerator design.

CONCLUSIONS

The continually changing waste stream implies a continuing need for adaptive waste management practices. The public role in managing MSW expanded dramatically during the 20th century in order to protect public health, and in doing so it displaced private loop-closing activities and imposed rigidities. The challenges for the new century are to improve the adaptability of the public management system and to reintegrate private sector and non-profit initiatives without increasing health or ecological risks. Industrial ecology provides a helpful systemic perspective and criteria for better waste management, but it offers less guidance on implementation issues.

This chapter has introduced concepts from the political economy perspective that are useful to those interested in implementing industrial ecology. Such tools clarify how society's collective decision-making mechanisms work, and with whom to interact when promoting change. The solid waste case illustrates that adding the 'who' to industrial ecological analysis increases its value.

45. Industrial ecology and integrated assessment: an integrated modeling approach for climate change

Michel G.J. den Elzen and Michiel Schaeffer

This chapter describes the background and modeling approaches used in simple climate models (SCMs) (Harvey *et al.* 1997). In general, SCMs are the simplified models used by the Intergovernmental Panel on Climate Change (IPCC) to provide projections of the atmospheric concentrations of greenhouse gases, global mean temperature and sea-level change response using as input emissions scenarios describing the future developments in the emissions of greenhouse gases. SCMs are computationally more efficient than more complex, computationally expensive three-dimensional models such as atmosphere–ocean general circulation models (AOGCMs). SCMs are therefore particularly suitable for multiple scenario studies, uncertainty assessments and analysis of feedbacks. The SCM approach is illustrated by applying one such model, meta-IMAGE (den Elzen 1998), to uncertainty analysis.

The SCM meta-IMAGE is a simplified version of the more complex climate assessment model IMAGE 2. IMAGE 2 aims at a more thorough description of the complex, long-term dynamics of the biosphere–climate system at a geographically explicit level ($0.5° \times 0.5°$ latitude–longitude grid) (Alcamo *et al.* 1996, 1998). Meta-IMAGE is a more flexible, transparent and interactive simulation tool that adequately reproduces the IMAGE-2.1 projections of global atmospheric concentrations of greenhouse gases, temperature increase and sea-level rise for the various IMAGE 2.1 emissions scenarios (see Figure 45.1). Meta-IMAGE consists of an integration of a global carbon cycle model (den Elzen *et al.* 1997), an atmospheric chemistry model and a climate model (upwelling-diffusion energy balance box model of Wigley and Schlesinger 1985 and Wigley and Raper 1992). The climate model also includes global temperature impulse response functions (IRFs; see, for example, Hasselmann *et al.* 1993) based on simulation experiments with various AOGCMs (den Elzen and Schaeffer 2001). This core model has lately been supplemented by a climate 'attribution' module, which calculates the regional contributions to various categories of emissions, concentrations of greenhouse gases and temperature and sea-level rise, especially developed for the evaluation of the Brazilian Proposal (UNFCCC 1997). Meta-IMAGE itself forms an integral part of the overall FAIR model (Framework to Assess International Regimes for burden sharing), which was developed to explore options for international burden sharing (den Elzen *et al.* 1999).

In the following section we describe a step-by-step approach along the cause and effect chain of climate change: from emissions to concentrations, from concentrations to radiative forcing and, finally, from radiative forcing to global mean surface air temperature

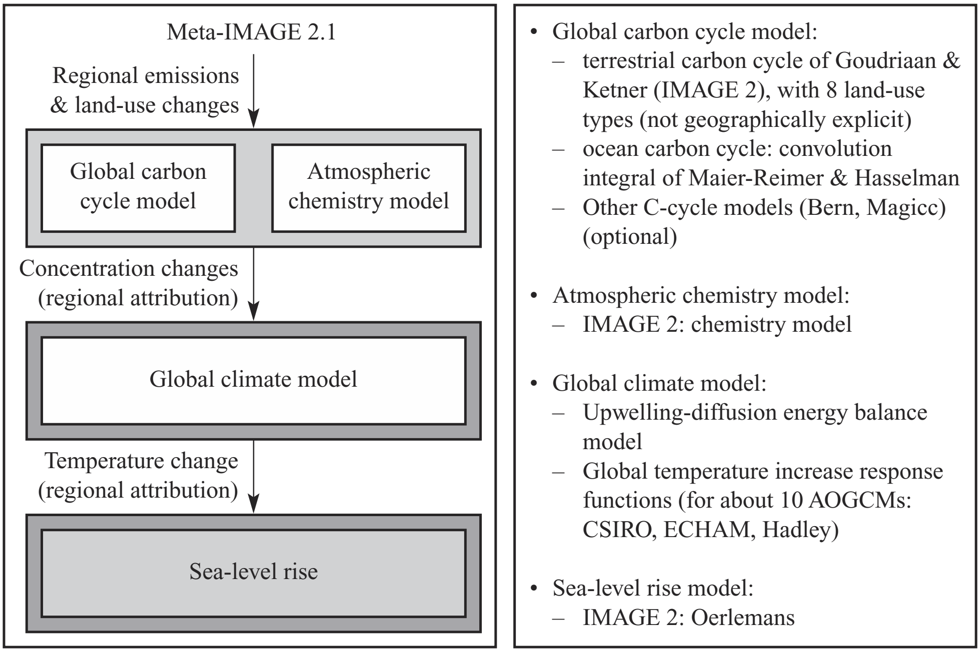


Figure 45.1 The climate assessment model meta-IMAGE 2.1 as used for the model analysis

increase. Various modeling approaches being used in SCMs, and in particular meta-IMAGE, are discussed. In the subsequent uncertainty analysis, we present an example of such a model analysis with meta-IMAGE of the impact of various carbon balancing mechanisms on the future projections of the atmospheric CO_2 concentration and the global mean temperature increase.

INTEGRATED MODELING APPROACH FOR CLIMATE CHANGE

From Emissions to Concentration

The concentration of a non-carbon dioxide greenhouse gas is modeled using a mass balance equation, in which the removal process in the stratosphere is proportional to the atmospheric concentration and inversely proportional to the atmospheric lifetime of a greenhouse gas. The same methodology is adopted in most current IPCC SCMs (Harvey *et al.* 1997). For nitrous dioxide (N_2O) and the halocarbons (including CFCs), a constant lifetime is used.

For methane (CH_4), matters are more complicated. Its chemical removal rate, and therefore its atmospheric lifetime, depend on the concentration of CH_4 itself. The latter is affected by the concentrations of other gases like NO_x, CO and VOCs. Therefore the lifetime of CH_4 is non-linearly dependent on the atmospheric composition. The lifetime of

CH_4 is time- and scenario-dependent and either the atmospheric chemistry has to be taken into account or the lifetime must be made time-dependent using previous results from (three-dimensional) chemical models. The current atmospheric lifetime is about nine years (Harvey *et al.* 1997). In addition to removal by chemical reactions in the atmosphere, CH_4 is also absorbed by soils, with a specific time constant of 150 years.

Regarding carbon dioxide (CO_2), there are considerable uncertainties in our knowledge of the present sources of, and sinks for, the anthropogenically produced CO_2. In fact, the only well-understood source is fossil fuel combustion while, in contrast, the source associated with land-use changes is less understood. The amount of carbon remaining in the atmosphere is the only well-known component of the budget. With respect to the oceanic and terrestrial sinks, the errors are likely to be in the order of ±25 per cent and ±100 per cent, respectively, mainly resulting from the lack of adequate data and from the deficient knowledge of the key physiological processes within the global carbon cycle (for example, Schimel *et al.* 1995). The uncertainties can be expressed explicitly in a basic mass conservation equation reflecting the global carbon balance (all components in gigatons of carbon content per year = GtC/yr):

$$\frac{dC_{CO2}}{dt} = E_{fos} + E_{land} - (S_{oc} + E_{for} + I), \tag{45.1}$$

where dC_{CO2}/dt is the change in atmospheric CO_2, E_{fos} is the CO_2 emission from fossil fuel burning and cement production, E_{land} the CO_2 emission from land-use changes, S_{oc} the CO_2 uptake by the oceans and E_{for} the CO_2 uptake through forest regrowth. To balance the carbon budget, a remaining term, I, which represents the missing sources and sinks, is introduced. I might therefore be considered as an apparent net imbalance between the sources and sinks. Analysis of the net imbalance in the global carbon cycle has become a major issue since the first IPCC scientific assessment report (IPCC 1990). Table 45.1 presents the global carbon balance over 1980–89 in terms of anthropogenically induced perturbations to the natural carbon cycle, as given by the IPCC (Schimel *et al.* 1995). Schimel *et al.* stated that the remaining imbalance of 1.3±1.5 GtC/yr might be attributable to terrestrial sink mechanisms; those are the terrestrial feedbacks of CO_2 and N fertilization and the temperature feedbacks on net primary production and soil respiration.

Table 45.1 Components of the global carbon dioxide mass balance, 1980–89, in terms of anthropogenically induced perturbations to the natural carbon cycle

Component (GtC/yr)	1980–89
Emissions from fossil fuel burning and cement production (E_{fos})	5.5 ± 0.5
Net emissions from land-use change (E_{land})	1.6 ± 1.0
Change in atmospheric mass of CO_2 (dC_{CO2}/dt)	3.3 ± 0.2
Uptake by the oceans (S_{oc})	2.0 ± 0.8
Uptake by northern hemisphere forest regrowth (E_{for})	0.5 ± 0.5
Net imbalance [$I = (E_{fos} + E_{land}) - (dC_{CO2}/dt + S_{oc} + E_{for})$]	1.3 ± 1.6

Source: Schimel *et al.* (1995).

The concentration of CO_2 is calculated in carbon cycle models on the basis of the mass balance as described in equation (45.1). The models consist of a well-mixed atmosphere linked to oceanic and terrestrial biospheric compartments. The oceanic component can be formulated as an upwelling-diffusion model (Siegenthaler and Joos 1992) or can be represented by a mathematical function (known as a convolution integral), which can be used to closely replicate the behavior of other oceanic models (Harvey 1989; Wigley 1991) such as in MAGICC (Wigley and Raper, 1992) and meta-IMAGE. The terrestrial component in both models is vertically differentiated into carbon reservoirs such as vegetation biomass, detritus, topsoil, deep soil and stable humus (Harvey 1989). For meta-IMAGE only, this component is also horizontally differentiated into eight land-use types: forests, grasslands, agriculture and other land for the developing and industrialized world (den Elzen 1998), which allows us to analyze the effect of land-use changes such as deforestation on the global carbon cycle.

To obtain a balanced past carbon budget in these models and therefore a good fit between the historical observed and simulated atmospheric CO_2 concentration, it is essential to introduce additional terrestrial sinks. Meta-IMAGE uses the CO_2 fertilization feedback and the temperature feedback on net primary production and soil respiration. Although the *N* fertilization feedback was included in the earlier version of the meta-IMAGE model (den Elzen *et al.* 1997), this feedback is, because of the consistency requirement with the IMAGE model, now excluded (den Elzen 1998). The parameterizations of these feedbacks have been derived from experiments with the IMAGE 2.1 model.

From Concentration to Radiative Forcing

Increased concentrations of greenhouse gases in the atmosphere lead to a change in the radiation balance. The basic effect is that the atmosphere becomes less transparent to thermal radiation. More heat is retained, although a number of climate feedbacks complicate this picture (Schimel *et al.* 1995). A good indicator for the change in radiation balance is radiative forcing. Radiative forcing is defined as the deviation from the pre-industrial radiative balance at the tropopause (border between troposphere and stratosphere) as a result of changes in greenhouse gas concentrations (while allowing the stratosphere to adjust to thermal equilibrium). This radiative forcing drives the changes in the free atmosphere and is the principal determinant for a change in surface air temperature. This results from the fact that the surface, planetary boundary layer and troposphere are so tightly coupled that they have to be treated as one thermodynamic system. The change in radiative balance at the tropopause then determines the change in energy input and outflow of that system.

Well-mixed gases (gases with a lifetime longer then the mixing time of the atmosphere) have a uniform concentration throughout the atmosphere. When using a global average for the vertical profile of temperature, water vapor and clouds, an assessment can be made of the global average radiative forcing response to an increase in the concentration of a particular greenhouse gas.

Each gas absorbs radiation in certain frequency intervals called ‘absorption bands’. If the concentration of a greenhouse gas is low, the troposphere will generally be transparent to radiation at the frequency of absorption of that gas. An increase of concentration leads to a practically linear increase in radiative forcing. This applies to the halocarbons, for example. As the concentration of a greenhouse gas increases, this dependence of forcing on concentration gradually 'saturates', as in the case of CO_2, CH_4 and N_2O. For these gases, the net flux at the tropopause at the frequency of strongest absorption is already close to zero. The change in net flux resulting from increased concentration will therefore also be small. Increased concentration will, however, still lead to increased absorption at the edges of the strongest absorption bands and at weaker bands. The saturation effect is strongest for CO_2 and somewhat less so for CH_4 and N_2O.

Another complication is that some greenhouse gases absorb radiation in each other's frequency domains. This is called 'the overlap effect' and is especially relevant for methane and nitrous oxide. Increases in CH_4 concentration decrease the efficiency of N_2O absorption and vice versa. The present SCMs like meta-IMAGE use the global radiative forcing dependencies of the IPCC (Harvey *et al.* 1997), including the major saturation and overlap effects.

From Radiative Forcing to Global Mean Temperature Increase

The large heat capacity of the oceans plays an important role in the time-dependent response of the climate system to external forcing, such as increased concentrations of greenhouse gases. Because the heat capacity of land surface and atmosphere is very small, ignoring inner ocean response would mean that, after a disturbance, the climate system would settle into a new equilibrium within a few years. However, heat is transported from the rapidly adjusting mixed (upper) layer of the ocean to deeper layers. This heat is therefore not available to warm the surface layer. Sea surface temperatures will rise more slowly. As a result, surface air temperatures over the ocean *and* land also increase more slowly. The time needed for the coupled atmosphere–ocean system to adjust fully to disturbances is extended.

Hasselmann *et al.* (1993) have shown that the time-dependent temperature response of coupled AOGCMs can be very well described using a linear combination of exponential decay terms, called impulse response functions (IRFs). As explained in Hasselmann *et al.* (1993), IRFs form a simple tool to describe ('mimic') mathematically transient climate model response to external forcing. A two-term IRF model as used here (as in Hasselmann *et al.* 1993) is based on the following convolution integral, relating global mean temperature response ΔT to time-dependent external forcing $Q(t)$:

$$\Delta T(t) = \frac{\Delta T_{2\times}}{Q_{2\times}} \int_{t_0}^{t} Q(t') \left(\sum_{s=1}^{2} l_s \frac{1}{\tau_s} e^{\frac{t-t'}{\tau_s}} \right) dt', \tag{45.2}$$

where $Q_{2\times}$ is the radiative forcing for a doubling of CO_2 and $\Delta T_{2\times}$ is the climate sensitivity, that is, the long-term (equilibrium) annual and global mean surface air temperature increase for a doubling of CO_2 concentration. The climate sensitivity is an outcome of all geophysical feedback mechanisms and their associated uncertainties, and is lying within the range of 1.5 to 4.5 °C, with a 'best guess' value of 2.5 °C; l_s is the amplitude of the 1st or 2nd component with exponential adjustment time constant τ_s, while $l_1 + l_2 = 1$. One of the two terms of these IRFs describes the rapid response and the other the slower response. The rapid response part dominates the response on a time scale of a few decades. This means effectively that the slower response part is of little relevance if the policy horizon only extends over a few decades. Still, a significant part (roughly 50 per cent) of the final global warming will manifest itself decades to centuries later. Called the 'warming commitment', this is also what causes sea-level rise to continue long after stabilization of greenhouse gas concentrations. The heat transport processes discussed above determine the balance between the fast and slow adjustment terms. This balance is still a source of uncertainty. Mathematically, the two-term IRF approach is equivalent to a two-box energy balance model. However, these boxes respond independently to the forcing. This makes it impossible to readily link the different IRF terms to specific elements in the physical climate system, such as mixed layer and deeper ocean.

Therefore the more physically based approach of the linked multi-box energy balance climate model is more often applied in SCMs. The original version of this type of model is described in Wigley and Schlesinger (1985) and Wigley and Raper (1992), although it has been modified since to include different climate sensitivities for land and oceans, and a variable ocean upwelling rate. The basic heat balance equation is described as follows:

$$C_m \frac{d\Delta T_o(t)}{dt} = Q(t) - \lambda \Delta T_o(t) - \Delta F(t), \tag{45.3}$$

where C_m is the heat capacity of the ocean mixed layer (Watts per year over degrees C times meters squared $= Wm^{-2}{}^\circ C^{-1}$), $Q(t)$ the total radiative forcing (W/m^2), ΔT_o the change in temperature of the ocean mixed layer (°C), ΔF the change in heat flux from the mixed layer to deeper ocean layers (W/m^2) and 1 the climate sensitivity parameter ($Wm^{-2}{}^\circ C^{-1}$); that is, $Q_{2\times} / \Delta T_{2\times}$, with: $Q_{2\times}$ being the radiative forcing for a doubled atmospheric CO_2 concentration ($= 4.37\ W/m^2$; Harvey *et al.* 1997). ΔF is calculated from the diffusion parameter and the transport terms.

As a reference climate model in meta-IMAGE, we use the upwelling-diffusion box climate model as described above, with a default climate sensitivity of $\Delta T_{2\times} = 2.35\,^\circ C$ to match IMAGE 2.1 results. For alternative temperature calculations, we also use a set of IRFs, with parameter settings derived from a range of AOGCMs (den Elzen and Schaeffer 2001).

From Global Mean Temperature Increase to Sea-level Rise

Global warming is expected to cause a sea-level rise due to ocean expansion, the melting of glaciers and ice caps, and the changes in the volume of the Greenland and Antarctic ice sheets. These processes are represented in the model by simple dynamic relationships presented by Oerlemans (1989) and Alcamo *et al.* (1998), using the parameterizations of den Elzen (1998).

MODEL EXPERIMENTS

The following uncertainty analysis presents an application of the integrated climate change cause-and-effect chain model described in the previous section. In this analysis several carbon cycle model parameters are adjusted to let the model's carbon cycle output 'browse' the uncertainty range of observed carbon budget components (Table 45.1). We then compare this uncertainty range with the uncertainty in climate system response. Finally, the synergy between these two major sources of uncertainty in climate change assessments is highlighted.

Input Data

The main input data of meta-IMAGE are composed of the anthropogenic emissions of the greenhouse gases and the land-use changes. The fossil fuel CO_2 emissions for the period

1751–1995 are based on the CO_2 emissions database of the Oak Ridge National Laboratory (ORNL) – CO_2 Data and Information Assessment Center (CDIAC) (Marland and Rotty 1984; Marland *et al.* 1999; Andres *et al.* 1999). The ORNL–CDIAC emission data are limited to CO_2 emissions from fossil fuels and cement production. The historical anthropogenic emissions of the other non-CO_2 greenhouse gas emissions and SO_2 are therefore taken from the EDGAR (Emission Database for Global Atmospheric Research) data set (Olivier *et al.* 1996) and the HYDE database (Klein-Goldewijk and Battjes 1997). The 1995–2100 anthropogenic emissions of all greenhouse gases and SO_2 are based on the IMAGE 2.1 Baseline-A scenario (Alcamo *et al.* 1998). This forms a 'business as usual' scenario, one without implementation of explicit climate change mitigation policies. The historical CO_2 emissions from land-use changes are based on Houghton and Hackler (1995) and Houghton *et al.* (1987). The land-use changes, important for the terrestrial carbon cycling processes, are also based on Houghton and Hackler (1995), but further disaggregated in the four major land cover types: forests, grasslands, agriculture and other land, for the developing and industrialized world, as used in meta-IMAGE. The area changes for the above four land-use categories for the developing and industrialized world for the period 1990–2100 are also based on aggregated land-use data of the IMAGE 2.1 Baseline-A scenario. The same holds for the 1995–2100 regional CO_2 emissions from land-use changes.

Various Balancing Approaches for the Past Carbon Budget

A key principle in the model analysis here is that we define different assumptions for the terrestrial biogeochemical feedbacks in order to balance the past carbon budgets; and subsequently, future atmospheric CO_2 concentration projections are made. In the reference case, model parameters representing the key terrestrial feedbacks (that is, the CO_2 fertilization effect and temperature feedback on net primary production and soil respiration) and oceanic uptakes are set at the default values used within meta-IMAGE. This leads to a balanced past carbon budget and a good fit between the historically observed and simulated CO_2 concentrations. The simulated carbon fluxes of the components of the carbon budget for the 1980s (1980–89) are similar to the IPCC estimates (Table 45.2). In the sensitivity experiments the scaling factors for the ocean flux, emissions from land-use changes, northern hemispheric terrestrial uptake and CO_2 fertilization feedback parameters are varied so that all parameter combinations will lead to a well-balanced past carbon budget. The temperature feedback parameters are kept at their default values. The simulated carbon fluxes of the carbon budget, that is, the oceanic and terrestrial carbon uptake and the CO_2 emissions from land-use changes during the 1980s, should be between the upper and lower boundaries of the IPCC estimates (Table 45.2). In the first two extreme 'oceanic uptake' cases, a high and a low oceanic uptake, the scaling factor for the ocean flux is set at the maximum and minimum values. This leads to the 1980s oceanic uptake of 3.0 and 1.0GtC/yr, respectively. The balanced past carbon budget is now achieved by scaling the CO_2 fertilization feedback parameters and the northern hemispheric terrestrial uptake. The terrestrial sink for the 1980s then varies between 0.8 and 2.3GtC/yr (Table 45.2). These two cases have been selected from other similar conditioned simulation experiments as two extreme examples of the balancing variations in the terrestrial and oceanic uptake fluxes (representing the terrestrial and oceanic uncertainties, in which the impact of variations in the temperature feedback parameters is ignored). The resulting projected CO_2 concentra-

Table 45.2 Components of the carbon budget (in GtC/yr), 1980–89, according to the IPCC (Schimel et al. *1995) and model simulations for the carbon balancing experiments*

Component	IPCC estimate	Reference case	High oceanic uptake	Low oceanic uptake	High deforestation	Low deforestation
CO_2 emissions from fossil fuel burning and cement production (E_{fos})	5.5 ± 0.5	5.5	5.5	5.5	0.0	5.5
CO_2 emissions from land-use changes (E_{land})	1.6 ± 1.0	1.6	1.6	1.6	2.6	0.6
Change in atmospheric CO_2 (dC_{CO2}/dt)	3.3 ± 0.2	3.3	3.3	3.3	3.3	3.3
Uptake by the oceans (S_{oc})	2.0 ± 0.8	2.0	3.0	1.0	2.0	2.0
Uptake by northern hemisphere forest regrowth (E_{for})	0.5 ± 0.5	0.5	0.0	0.5	0.5	0.5
Additional terrestrial sinks (IPCC: $[E_{fos} + E_{land}] - [dC/dt^* + E_{for} + S_{oc}]$)	1.3 ± 1.6	1.3	0.8	2.3	2.3	0.3

tion range in 2100 varies between 687 and 719ppmv, while the 2100 concentration is 717ppmv for the reference case (dark gray area in Figure 45.2(b)).

Further experiments consider the uncertainties in the CO_2 emissions from land-use changes (land-use sources and carbon sink uncertainties) by setting its scaling factor at maximum and minimum values (emissions during the 1980s between 0.6–2.6GtC/yr) (Table 45.2). The extreme upper and lower CO_2 concentration projections are now achieved (662 and 794 ppmv, respectively, by 2100), by balancing the past carbon budget with only the CO_2 fertilization effect, while the oceanic parameters are kept constant (light gray area in Figure 45.2(b)). Summarized, these experiments show that various balanced past carbon budgets in the model lead to a range in the 2100 atmospheric CO_2 concentration for the IMAGE Baseline-A scenario of about 10 per cent above and below the central projection of ~717ppmv.

Figure 45.2 shows the temperature increase for the Baseline-A scenario using the CO_2 concentration pathway from the reference case according to the meta-IMAGE climate model (solid line). In Figure 45.2(a), uncertainty of carbon cycle modeling is illustrated, given by uncertainty in terrestrial and oceanic carbon sink fluxes (dark gray), and in the sink and land-use sources (light gray). Figure 45.2(b) illustrates uncertainty in the climate system response as estimated by using different IRFs of AOGCMs. Dark gray: range of outcomes for IPCC's 'best guess' climate sensitivity of 2.5 °C combined with the IRF time scale parameters of the AOGCMs. Grey: IRF time scale parameters combined with their respective climate sensitivities. Light gray: IRF time scale parameters arbitrarily combined with climate sensitivities in the full IPCC range of 1.5–4.5 °C. The two uncertainty ranges represent the uncertainties in the terrestrial and oceanic carbon sink fluxes (dark gray), and in the sink and land-use sources (light gray).

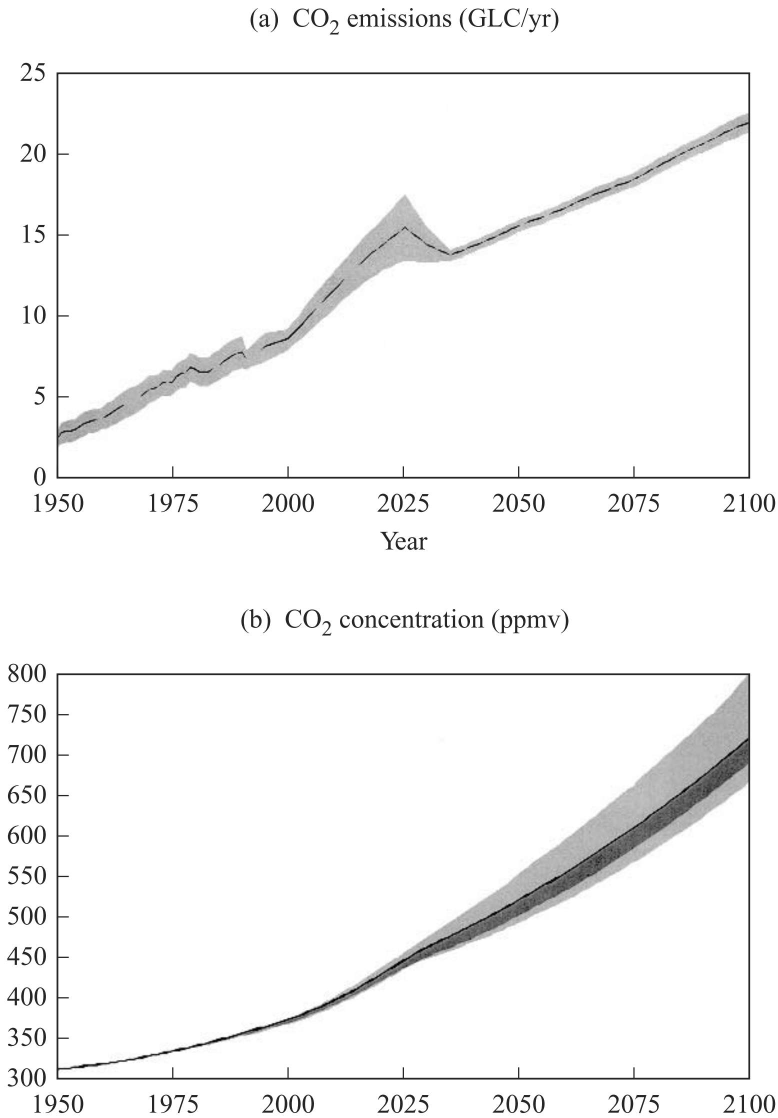
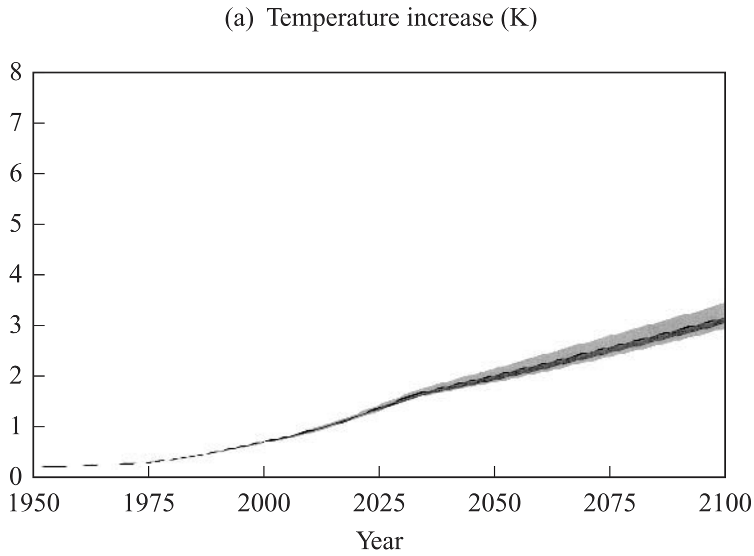


Figure 45.2 Global anthropogenic CO_2 emissions (a) and CO_2 concentrations (b) Baseline-A scenario according to the meta-IMAGE model for the carbon balancing experiments

Using Temperature Response Functions of Various AOGCMS

We now apply the results of a number of climate models to reflect uncertainty in climate modeling. In order to do so, we have diagnosed a range of different response time scale parameters and climate sensitivities from sophisticated atmosphere–ocean climate models in den Elzen and Schaeffer (2000). Using this range results in the spread in projected tem-



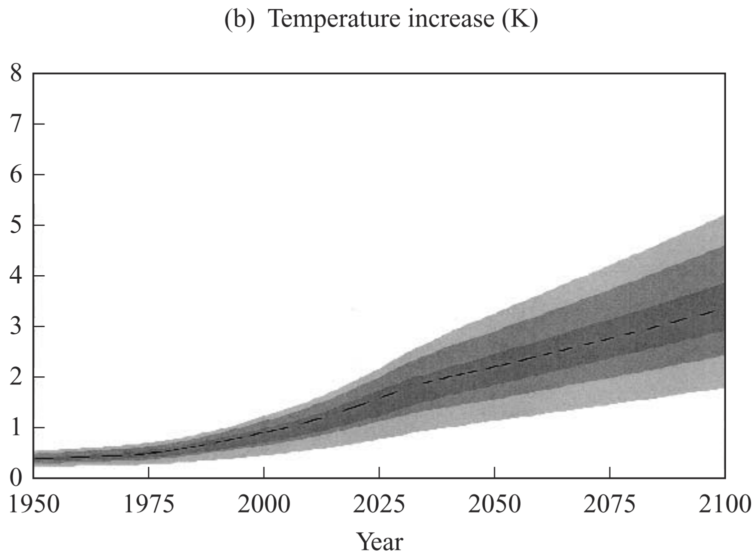


Figure 45.3 *Global anthropogenic CO_2 emissions (a) and CO_2 concentration pathway (b) from the reference case according to the meta-IMAGE model*

perature response shown in Figure 45.3 (gray area). Here the solid line represents the temperature response of our meta-IMAGE reference case, which is not necessarily the most likely. The global surface temperature increase for the reference projection is about 3.1 °C for the period 1751 (pre-industrial) to 2100 for the reference case. The innermost, dark gray area depicts the range of results if all the different IRF time scale parameters are applied using the same IPCC 'best guess' climate sensitivity of 2.5 °C. The uncertainty range broadens significantly if the IRF time scale parameters are combined with their respective climate sensitivities, ranging from 1.58 to 3.7 °C. Finally, the range broadens even further if the IRF time scale parameters are arbitrarily combined with climate sensitivities from the full IPCC range of 1.5–4.5 °C (light gray). Figure 45.3 clearly shows that the climate sensitivity plays a dominant role in determining the range of absolute temperature increase.

Figure 45.3(a) illustrates the uncertainty of carbon cycle modeling as shown by terrestrial and oceanic carbon sink fluxes (dark gray), and in the sink and land-use sources (light gray). Figure 45.3(b) illustrates uncertainty in the climate system response as estimated using different IRFs of AOGCMs. Dark gray: range of outcomes for IPCC's 'best guess' climate sensitivity of 2.5 °C combined with the IRF time scale parameters of the AOGCMs. Grey: IRF time scale parameters combined with their respective climate sensitivities. Light gray: IRF time scale parameters arbitrarily combined with climate sensitivities in the full IPCC range of 1.5–4.5 °C.

Overall Uncertainties of Global Carbon Cycle and Climate Models

Having analyzed the effect of some major uncertainties in the carbon cycle and climate in the preceding sections, we now compare their relative importance for projected temperature increase. In Figure 45.4, the central dark gray area depicts total uncertainty in the carbon cycle balancing exercise (compare with Figure 45.3(a), while the gray area reflects total uncertainty from climate modeling (compare with Figure 45.3(b)). Finally, the light gray area shows the resulting total range in outcomes using the overall uncertainties in the carbon cycle and climate. The extension of uncertainty on the upper side is worth noting;

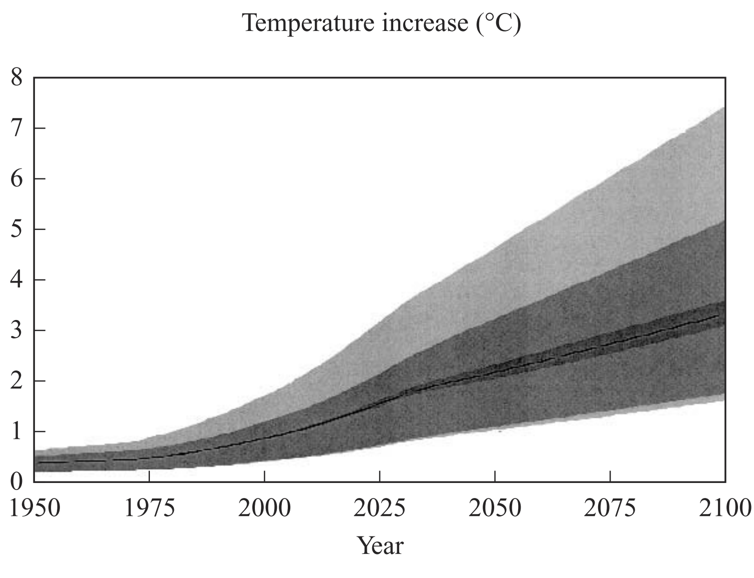


Figure 45.4 The global mean surface temperature increase for the Baseline-A scenario for the model uncertainties in the carbon cycle and climate models, and the combined effect of both

this is a result of the strong temperature feedback on soil respiration as it is triggered by a high temperature increase in meta-IMAGE when a high climate sensitivity is applied. Clearly, uncertainty in climate modeling is dominant in determining absolute temperature increase, caused by uncertainty in climate sensitivity. However, because of the temperature–respiration feedback, the 'synergy' of carbon cycle and climate system increases this uncertainty significantly. This forms a perfect illustration of the advantage of using an integrated modeling approach.

CONCLUSIONS

In this chapter we have described the cause-and-effect chain of climate change: from emissions and concentrations of greenhouse gases to global mean temperature and sea-level rise. We briefly treated the modeling approaches being used in SCMs and in particular the integrated assessment model meta-IMAGE. Although the model representations of the biological, physical and chemical processes in these models are highly aggregated representations of complicated, geographically dependent and only partly known processes, SCMs prove to be valuable tools for describing the climate change problem in a quantitative, adequate way. SCMs are particularly useful in providing climate projections in multiple scenario studies and uncertainty assessments, as was shown in the illustrative model analysis.

46. Earth systems engineering and management

Braden R. Allenby

Earth systems engineering and management (ESEM) is a new area of study arising from the confluence of several trends in different fields. As a result of the Industrial Revolution, the globalization of the Greco-Judaeo-Christian Eurocentric civilization and its technologies, and explosive growth in human population levels and economic activity, the dynamics of many fundamental natural systems (for example, the carbon, nitrogen, sulfur, phosphorus and hydrologic cycles; atmospheric and oceanic systems; the biosphere at scales from the genetic to the species and community levels) are now dominated by the activities of one species – ours (Turner *et al.* 1990; Ayres *et al.* 1994; Nriagu 1994; Smil 1997; Vitousek *et al.* 1997). The Earth as it now exists increasingly reflects the perhaps unintended and unconscious, but nonetheless real, design of a single species. Although this process has been accelerated by the Industrial Revolution, 'natural' and human systems on all scales have in fact been affecting each other, and evolving together, for millennia, and they are now more tightly coupled than ever. Copper production in China during the Sung Dynasty, as well as in Athens and the Roman Republic and Empire, are reflected in deposition levels in Greenland ice (Hong *et al.* 1996). And lead production in ancient Athens, Rome and medieval Europe is reflected in increases in lead concentration in the sediments of Swedish lakes (Renberg *et al.* 1994). The build-up of carbon dioxide in the atmosphere began, not with the post-World War II growth in consumption of fossil fuel, but with the deforestation of Europe, Africa and Asia over the past centuries and millennia (Jager and Barry 1990). Humanity's impacts on biota, both directly through predation and indirectly through the introduction of new species to indigenous habitats and shifts to agricultural and urban technologies and cultural patterns, has been going on for centuries as well (Jablonski 1991; Diamond 1997; Redman 1999).

This observation is neither new nor unique: it was powerfully expressed over a hundred years ago in the classic *Man and Nature* by George Perkins Marsh (1973 [1864]). See also Thomas (1956a) and Turner *et al.* 1990. William Clark noted over a decade ago, in a 1989 special issue of *Scientific American* entitled 'Managing Planet Earth', 'Self-conscious, intelligent management of the earth is one of the great challenges facing humanity as it approaches the 21st century.' Gallagher and Carpenter (1997, p.485) introduced a special issue of *Science* on human-dominated ecosystems by noting that the cultural construct of 'pristine' or 'natural' ecosystems untouched by human activity is 'collapsing in the wake of scientists' realization that there are no places left on Earth that don't fall under humanity's shadow' (*Science* 1997).

The evolution of industrial ecology has also played a role in supporting the development of ESEM. Industrial ecology tools such as materials flow analysis (Chapter 8), input–output models (Chapter 10), life cycle assessment (Chapter 12), integrated assessment (Chapter 45) and design for environment (Chapter 35), have all contributed to a greater understanding of the linkages between technology systems, cultural patterns and natural systems (Grübler 1998). In fact, the first use of the term 'earth systems engineering' was in the *Journal of Industrial Ecology* (Allenby 1999b, where the first dialog on the concept also was published; Friedman 1999; Allenby 1999c). And the second Gordon Conference on Industrial Ecology, held in New Hampshire in June 2000, focused on the concept of earth systems engineering and management. Reflecting perhaps the productive ferment in the field of industrial ecology, the question of whether ESEM is properly considered a part of industrial ecology, or a new area of study which is based in important ways on industrial ecology, nevertheless remains open for many industrial ecologists.

An additional important area of study involves the work that has been done over the past decade or so looking at geo-engineering options to mitigate global climate change (Keith and Dowlatabadi 1992; USNAS 1992, especially ch. 28, pp. 433–64; Rubin *et al.* 1992). Proposals in the traditional geo-engineering area revolve primarily around mechanisms for reducing incident solar radiation, including placing mirrors, reflectors or particles in space between the Sun and the Earth (Teller *et al.* 1997; Govindasamy and Caldeira 2000). Reflecting the nascent state of both areas of study, the line between these geo-engineering proposals and ESEM is not yet clear: recent proposals to reduce atmospheric carbon dioxide levels by fertilizing plankton growth in the ocean by applying iron (Kerr 1994; Behrenfeld *et al.* 1996; Coale *et al.* 1996) or to dam the Strait of Gibraltar to mitigate shifts in oceanic circulation patterns resulting from climate change (Johnson 1997) appear to lie somewhere in the middle. Current research to prove the validity of systems to capture carbon dioxide emissions from fossil-fueled power plants and sequester them in underground geologic formations or aquifers, thus supporting a hydrogen/electric energy system that would not increase global climate change forcing, would appear to be closer to ESEM (Herzog and Drake 1996; Socolow 1997; Allenby 1999b).

As these examples illustrate, the global climate change negotiations taken as a whole are perhaps the classic current case study in ESEM, although the broad, multidisciplinary, systems-based approach which should characterize any ESEM activity (just as it does industrial ecology) is lacking (Allenby 2000b). But there are other examples as well: these include the continuing efforts to manage the Baltic Sea and Everglades, managing regional forests to make them sustainable, exploitation of local and regional fisheries, and, of course, continued challenges from invasive species, especially those introduced during great human migrations such as the Polynesian or European episodes (Gunderson *et al.* 1995; Diamond 1997; Berkes and Folke 1998; Kaiser 1999; Redman 1999).

With this as background, ESEM may be more formally defined as 'the capacity to rationally engineer and manage human technology systems and related elements of natural systems in such a way as to provide the requisite functionality while facilitating the active management of strongly coupled natural systems' (Allenby, 2000/2001). Important elements of this definition include treating human and natural systems as coherent complexes, to be addressed in a unified manner, and the understanding that requisite functionality includes not just the desired output of the technological system – energy, for example – but respect for, and protection of, the relevant aspects of coupled natural systems: aesthetics, ecosystem services such as flood control and biodiversity as an independent value, and others.

One important difference between 'traditional' engineering disciplines and ESEM is worth emphasis. While virtually all engineering activities to some extent reflect the cultures within which they are embedded, the cultural dimensions of traditional engineering are often not critical to the impacts of the engineering activity. Designing a toaster does indeed reflect some cultural dimensions, but understanding the roots of the western technological discourse is hardly a necessary component of the activity (just as it is not for a design for environment analysis of a particular manufacturing technique, for example). But the scale and scope of ESEM, and the critical existential importance of the systems with which ESEM deals, means that one must comprehend not just the scientific and technological domains, but the social science domains – culture, religion, politics, institutional dynamics – as well. The human systems implicated in ESEM are extraordinarily powerful, with huge inertia and resistance to change built into them, and ESEM will fail as a response to the conditions of our modern world unless these are respected and understood.

The evolution of Eurocentric Judaeo-Christian capitalist and technology systems has swept the globe (Landes 1998; Harvey 1996; Diamond 1997; Noble 1998). Relevant implications of this historical process include commoditization of the world, including nature, which in developed countries is increasingly purchased at stores in up-market malls, in theme parks (reflecting not 'natural' ecological dynamics but late 20th-century ideology, including, of course, corporate sponsorship), and as eco-tour packaged 'experiences' (Marx and Engels, (1998 [1847]); Cronon 1996). The process of commoditization, the globalization of culture through movements such as post-modernism (Harvey 1996; Anderson 1998) and urbanization have had several fundamental effects. First, the cultural concepts of 'nature', 'wilderness' and related terms are changing for many people. Second, the complexity of society, and of the couplings between human and natural systems, is increasing radically (Harvey 1996; Redman 1999; Allenby 2000/2001). Finally, this increased complexity is reflexively affecting the governance structures within which environmental and technological issues have traditionally been addressed: the absolute primacy of the nation-state is being replaced by a far more fluid, and complex, dynamic structure involving a number of stakeholders, including private firms, NGOs and communities of all kinds (Chapter 6).

Another fundamental issue raised by ESEM is the pivotal role of values: ESEM by its nature is a means to an end which can only be defined in ethical terms. Simply put, the question, 'To what end are humans engineering, or *should* humans engineer, the Earth?' is a moral and religious matter, not a technical one. Moreover, as with ESEM itself, it is not hypothetical: human institutions are implicitly answering that question every day, and thus positing an answer. The climate change negotiations, for example, embed a number of ethical issues, but they tend to be treated implicitly, not explicitly, and thus are both unrecognized and, frequently, substantial unrecognized barriers to progress (Allenby 2000b). In general, failure to recognize the strong coupling between human activity on large scales and the state of environmental systems has to date permitted the engineering process to proceed without explicit consideration of the ethical content of the results. At some point this veil of ignorance will be pierced, perhaps in addressing the complex issues raised by global climate change and possible mitigation. Currently, however, this process is in its early stages.

It is clear that the institutional, ethical and knowledge bases necessary to support

ESEM as an operational field do not yet exist (and may never exist, in the view of skeptics). Nevertheless, experience to date with complex systems engineering projects (Pool 1997; Hughes 1998), international efforts to manage stratospheric ozone depletion, loss of biodiversity and habitat, and global climate change, and 'adaptive management' practices regarding complex natural resource systems such as the Everglades or the Baltic Sea (Gunderson *et al*. 1995; Berkes and Folke 1998) can be assessed to generate a basic set of ESEM principles. These can be roughly sorted into three categories: theory, governance, and design and engineering.

THEORY

The theoretical underpinnings of ESEM reflect several important realities, particularly the scope and scale of the systems involved, and our current levels of (often underappreciated) ignorance. Accordingly, perhaps the most important principle of all is precautionary.

- Only intervene when necessary, and then only to the extent required. This principle is similar to the medical admonition, 'first, do no harm', and is based on the same rationale: when faced with a complex, uncertain, inherently unpredictable system, minimal interventions reduce the probability and potential scale of unanticipated undesirable system responses.
- At the ESEM level, projects and programs are not just scientific and technical in nature, but unavoidably have powerful economic, political, cultural, ethical and even religious dimensions. An ESEM approach should integrate all these factors.
- Unnecessary conflict surrounding ESEM projects and programs can be reduced by recognizing the difference between social engineering – efforts to change cultures, values or existing behavior – and technical engineering. Both need to be integrated in ESEM projects, but they are different disciplines and discourses, involving different issues and world views and, while both part of ESEM, should not be conflated.
- It follows from the above principles that ESEM requires a focus on the characteristics and dynamics of the relevant systems as systems, rather than just as the constituent artifacts. The artifacts will, of course, have to be designed in themselves as well; ESEM augments, rather than replaces, traditional engineering activities. This systems-based approach is, of course, a fundamental element of industrial ecology as well.
- Boundaries around ESEM initiatives should reflect real-world couplings and linkages through time, rather than disciplinary or ideological simplicity. It cannot be overemphasized that ideology, whether explicit or implicit, inevitably is a (frequently inappropriate and dysfunctional) oversimplification of the systems at issue and their dynamics, and, therefore, such approaches should be avoided to the extent possible.
- Major shifts in technologies and technological systems should be evaluated before, rather than after, implementation of policies and initiatives designed to encourage them. Thus, for example, encouraging reliance on biomass plantations as a global

climate change mitigation effort should not become national or international policy until predictable implications – further disruption of nitrogen, phosphorus and hydrologic cycles, for example – are explored (Allenby 2000b).

GOVERNANCE

Many authors have commented on obvious changes in the global governance system, which is migrating away from a model where the nation-state is sovereign, to one characterized by complex interactions among private firms, NGOs, communities, nation-states and other interest groups (see Chapter 6 in this volume). These changes, combined with the complexity of human and natural systems, give rise to a second category of principles involving ESEM governance.

- ESEM initiatives by definition raise important scientific, technical, economic, political, ethical, theological and cultural issues in the context of an increasingly complex global polity. Given the need for consensus and long-term commitment, the only workable governance model is one that is inclusive, democratic, transparent and accountable.
- ESEM governance models, which deal with complex, unpredictable systems, must accept high levels of uncertainty as endogenous to the discourse. ESEM policy development and deployment must be understood as a continuing dialog with the relevant systems, rather than a definitive endpoint. ESEM governance structures should accordingly place a premium on flexibility, and the ability to evolve in response to changes in system state and dynamics, and recognize the policy maker as part of an evolving ESEM system, rather than an agent outside the system guiding or defining it.
- Continual learning at the personal and institutional level must be built into the process, as is the case now in 'high reliability organizations' such as aircraft carrier operations or well-run nuclear power plants (Pool 1997). This learning process is messy and highly multidisciplinary, but it is particularly critical with ESEM projects, which in many cases will involve significant experimentation and a highly interactive relationship with the systems at issue.
- There must be adequate resources available to support both the project and the science and technology research and development which will be necessary to ensure that the responses of the relevant systems are understood.

DESIGN AND ENGINEERING

Finally, there are a set of principles that inform the design and engineering of ESEM systems.

- Know from the beginning what the desired and reasonably expected outcomes of any intervention are, and establish quantitative metrics by which progress may be tracked. Additionally, predict potential problematic system responses to the extent possible, and identify markers or metrics by which shifts in the probability of their occurrence may be tracked.

- Unlike simple, well-known systems, the complex, information-dense and unpredictable systems that are the subject of ESEM cannot be centrally or explicitly controlled. Rather than being exogenous to a system, the earth systems engineer will have to see herself or himself as an integral component of the system itself, closely coupled with its evolution and subject to many of its dynamics, which will require an entirely different psychology of engineering.
- Whenever possible, engineered changes should be incremental and reversible, rather than fundamental and irreversible. In all cases, scaling up should allow for the fact that, especially in complex systems, discontinuities and emergent characteristics are the rule, not the exception, as scales change. Locking in of inappropriate or untested design choices as systems evolve over time should be avoided.
- An important goal in earth systems engineering projects should be to support the evolution of resiliency, not just redundancy, in the system. Moreover, inherently safe systems are to be preferred to engineered safe systems. An inherently safe system, when it fails, fails in a non-catastrophic way; an engineered safe system is designed to reduce the risk of catastrophic failure, but there is still a finite probability that such a failure may occur.

At this point, it is apparent that the science and technology, institutional and ethical infrastructures necessary to support ESEM on a significant scale are not yet developed. The issue is not, however, whether the Earth will be engineered by the human species: that has been occurring and will continue to occur. Indeed, the Kyoto process is an example of incipient ESEM, although it is not yet a conscious or systematic activity. The real issue is whether humans can, and will, improve their management of the Earth rationally, intelligently and ethically. The development of ESEM, informed by industrial ecology, is an important step in developing that capability.

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