Gjalt Huppes
Masanobu Ishikawa
Editors

ECO-EFFICIENCY IN INDUSTRY AND SCIENCE 22

Quantified Eco-Efficiency

An Introduction with Applications

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Quantified Eco-Efficiency
An Introduction with Applications

Edited by

Gjalt Huppes
CML, Department of Industrial Ecology,
Leiden University, Leiden, The Netherlands

and

Masanobu Ishikawa
Graduate School of Economics,
Kobe University, Kobe, Japan

Springer

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Preface

This volume results from the work for the first and second conference on quantified eco-efficiency analysis for sustainability. Most papers are based on work presented at the first conference. However the general introduction to eco-efficiency, in chapter 1, reflects the enormous experience gained in the process of editing the diverse papers resulting from the conference.

As several papers show, eco-efficiency analysis is developing fast into a mature method of analysis, with a broad domain of applications. On methods, we selected two papers. One - presented at the conference but now detailed on the basis of further work - refers to the Maximum Abatement Cost method, see chapter 2. It avoids to a large extent value choices and assumptions, as on the validity of neo-classical assumption on the welfare meaning of stated private preferences regarding environmental effects. Using actual cost of emission reduction, it indicates domains of efficient and inefficient environmental improvements in projects with multiple environmental impacts. Some further empirical work on the integration towards s a single environmental indicator uses a shadow price method, applied at the level of LCA type midpoint impact categories, in chapter 4. A deviating voice is on the fundamentals of the eco-efficiency concept, in chapter 3. Are we using the right concepts; might other concepts, better linked to a natural science practice of technical input-output efficiency not be more fruitful for sustainability analysis in the long term?

The applied analysis covers the domains of agriculture, industry, products and consumption, and recycling, with cases as small as single products and technologies and as broad as regional recycling activities or large scale reforestation plans. In agriculture, land conversion schemes, especially reforestations plans are analysed using regional input-output analysis, see chapter 5. The industrial cases refer to supply chain management in furniture for improved eco-efficiency, in chapter 6, and the galvanising industry in chapter 7. An eco-efficiency based social procedure is designed in chapter 8 using cost advantages of environmental improvements to finance further improvements. Cases on products and consumption range from broad analysis of households performance, in chapter 9; to a specific product with high eco-efficiency potential, the smart window, in chapter 10; to methods for extending product life time through upgradeable product design, in chapter 11. Finally, there is a focus on recycling, one general paper in chapter 12 for selective promotion of secondary materials use at a general policy level, and one on advanced methods and technologies for regional plastics recycling (chapter 13).

The introduction to quantified eco-efficiency analysis, in chapter 1, reflects the development of ideas since the first conference. Parts of it have been published in the special issue on eco-efficiency of the Journal of Industrial Ecology. We thank the Journal for allowing us to take over sections from three papers there. The shift in content since then has been to better indicate the relation between macro level sustainability, as quality of the environment combined with global economic growth, and the micro-level of choices on products, technologies and policies, which through eco-efficiency analysis can better be geared to macro-level sustainability. There is direct relevance for eco-innovation and for general sustainability policy, aligning and integrating policies for economic development with environmental policies. This work surely is not finished yet and deserves broader attention in the sustainability analysis community.

Finally, we would like to thank EBARA Company from Japan who not only are the main sponsor of the conferences on Quantified Eco-Efficiency Analysis for Sustainability, but also are the sponsor of this book. We owe them gratitude for the unselfish advancement of sustainability science, from a truly global perspective.

The editors
Gjalt Huppes
Masanobu Ishikawa

Corresponding Authors

Fresner, Johannes (ch. 7)
STENUM GmbH
Geidorfgürtel 21, 8010 Graz, Austria
E-mail j.fresner@stenum.at

Harmelen, Toon van (ch. 4)
TNO Institute of Environment, Energy and Process Innovation
P.O. Box 1, NL-7300 AH Apeldoorn, Netherlands
E-mail toon.vanharmelen@tno.nl

Heijungs, Reinout (ch. 3)
CML, Department Industrial Ecology, Leiden University
P.O. Box 9518, NL-2300 RA Leiden, Netherlands
E-mail heijungs@cml.leidenuniv.nl

Huppes, Gjalt (editor, preface, ch. 1)
CML, Department Industrial Ecology, Leiden University
P.O. Box 9518, NL-2300 RA Leiden, Netherlands
E-mail huppes@cml.leidenuniv.nl

Indrianti, Nur (ch. 12)
E-mail ajisari@indo.net.id

Ishikawa, Masanobu (editor, preface, ch. 1)
Graduate School of Economics, Kobe University
2-1 Rokkodai-cho, Nada-ku, Kobe 657-8501, Japan
E-mail masanobu@yhc.att.ne.jp

Michelsen, Ottar (ch. 6)
Department of Industrial Economics and Technology Management
Norwegian University of Science and Technology (NTNU)
NO-7491 Trondheim, Norway
E-mail ottar.michelsen@iot.ntnu.no

Oka, Tosihiro (ch. 2)
Graduate School of Economics and Administration, Fukui Prefectural University
4-1-1 Kenjojima Matsuoka-cho Yoshida-gun, Fukui 910-1195, Japan
E-mail oka@fpu.ac.jp

Papaefthimiou, Spiros (ch. 10)
Solar Energy Laboratory, Physics Department, University of Patras
Patras 26500, Greece
E-mail spapaef@physics.upatras.gr

Spindler, Ernst-Josef (ch. 8)
Vinnolit GmbH & Co. KG
D 84489 Burghausen, Germany
E-mail ernst.spindler@vinnolit.com

Watanabe Kentaro (ch. 11)
Research into Artifacts, Center for Engineering, University of Tokyo
Komaba 4-6-1, Meguro-ku, Tokyo, Japan
E-mail nabeken@race.u-tokyo.ac.jp

Wier, Mette (ch. 9)
AKF, Institute of Local Government Studies
Nyropsgade 37, DK-1602 Copenhagen, Denmark
E-mail mw@akf.dk

Yabar, Helmut (ch. 13)
Div. of Sustainable Energy and Environmental Engineering
Graduate School of Engineering, Osaka University
2-1 Yamadaoka, Suita, Osaka 565-0871, Japan
E-mail yabar@ecolonia.env.eng.osaka-u.ac.jp

Zhang, Fan (ch. 5)
Harvard University
79 John F. Kennedy Street, Cambridge, MA 02138 USA
Email Fan_Zhang@ksgphd.harvard.edu

1 An introduction to quantified eco-efficiency analysis

Gjalt Huppes[a] and Masanobu Ishikawa[b]
[a] *CML, Department of Industrial Ecology, Leiden University, Netherlands*
[b] *Graduate School of Economics, Kobe University, Kobe, Japan*

1.1 The challenge of sustainability

A growing global population with growing affluence may well lead to reduced environmental quality and a diminishing quality of nature, ultimately jeopardizing the quality of human life and even human life itself. The challenge we face is to reduce the environmental consequences of our actions so as to reduce environmental risks and to retain the quality of the environment not only as is necessary for survival but also reflecting higher order values on nature and human life, as for example reflected in the concept of sustainability. The challenge has down to earth properties. Environmental impacts per unit of welfare, as eco-efficiency, on average should be appropriate for sustainability. Simplifying the analysis a bit, as by disregarding non-linearities and dynamics, in any year the total amount of environmental impacts should be within limits as set by sustainability considerations. In any one year this total amount is the sum-total of all micro-level economic actions in production, consumption and waste management, including investments and public sector activities. These economic actions grow in real terms, for the decades to come may be by four percent per year. Therefore, the eco-efficiency requirements on global society as a whole somehow have to be matched by eco-efficiency requirements on all our activities to counteract such expected growth. They should be reflected in all our economic decisions.

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 1–38.

There is no direct correspondence to micro level actions and decisions, however. The average macro level environmental burden per unit of expenditure, actual or allowable, cannot be matched with environmental impact per unit of value added in a micro level activity. Some activities by nature can be virtually without environmental impact, as in many cultural events like singing classes or mathematics studies. Other activities, like international travel and coal mining, have a high impact per unit of value added by shear technical necessity. Putting the same eco-efficiency requirement on all activities as is valid on average at a macro level clearly is not possible. Still, in order to reach the required eco-efficiency at macro level, that is the sum of our micro level actions in terms of value and environmental impact, there should be requirements on individual activities and decisions. Before engaging in the difficult and by nature political process of *who should do what* to safeguard our future, we first should know the empirical facts and developments: what is the eco-efficiency of our current activities, how do they develop and which options for further improving eco-efficiency do we have. Though statistically the macro level just is the sum of all micro level activities, the link to decision making is not so clear. Individual economic actions are not created in a void but are intricately related. Reducing emissions at one spot may well lead to more than compensating increases in other spots, as might be the case with bio-ethanol from grain in gasoline (Farrell et al. 2006). Production and consumption chains, and their waste management requirements, are intricately related and may cover many years as with investment goods and durable consumer goods. So the unit of decision making cannot just be individual activities, as their interrelations have to be taken into account. This leads to modeling of interrelations in developing product systems, of firms behavior, of regions and countries, with all complex feedback loops as are present in society. Eco-efficiency analysis of decisions depends on such models, simple or complex. Only for monitoring purposes, eco-efficiency analysis does not pertain to effects of decisions and actions, but to the environmental impacts and value created by activities which can be added to a yearly total for the world. The units to add up ultimately are single activities, however defined, and aggregates of these. Examples are firms; private consumption households; sectors; regions; and product systems. Only for very simple models the link to the macro level is relatively direct, as with environmentally extended input-output analysis. Also steady state type LCA is close, related to the requirement that the sum of all parts is the total, but still abstracts from the fact that actual product system cover a time span of many years or even decades.

As a next step, still not really normative and political, we may ask questions on optimality. Can we distinguish options as being either superior or inferior, as when a product is dominant in decision making terminology, having the same functionality with lower cost and with lower environmental impact than also available alternatives? Limited value judgments may suffice here. A choice for an option with the same value and lower environmental impact, or higher value with same environmental impact, is generally to be preferred. Based on such minimal Pareto-like assumptions, quantified eco-efficiency analysis already can give guidance on choices regarding technology and consumption. As one part of the macro equation, population growth, is hardly amenable to policy, at least not in the time frame of decades, and as rising global affluence is globally accepted as a central aim of public policies, we will have to look at the other part of the equation, the environmental impact per unit of welfare, shifting technologies of production and consumption in a sustainable direction. This discussion is has the same background as the Factor X discussion, but focuses on the precise nature of what constitutes the nominator and denominator in this Factor. That question boils down to how to measure eco-efficiency empirically and how to measure its development.

So, in this paper eco-efficiency analysis firstly is an instrument for sustainability analysis, primarily indicating an empirical relation in economic activities between environmental cost or value and environmental impact. This empirical relation can be matched against normative considerations as to how much environmental quality or improvement society would like to offer in exchange for economic welfare, or what the trade-off between the economy and the environment should be if society is to realize a certain level of environmental quality. Its relevance lies in the fact that relations between economy and environment are not self-evident, not at a micro level and not at the macro level resulting from micro-level decisions for society as a whole. Clarifying the why and what of eco-efficiency is a first step towards decision support on these two aspects of sustainability. With the main analytic framework established, filling in the actual economic and environmental relations requires further choices in modeling. Also, the integration of different environmental effects into a single score requires a clear definition of approach, because several partly overlapping methods exist. Some scaling problems accompany the specification of numerator and denominator, which need a solution and a certain amount of standardization is required before eco-efficiency analysis can become more widely used. With a method established, the final decision is how to embed it in practical decisionmaking. In getting the details of eco-efficiency better specified, its strengths, but also its weaknesses and limitations, need to be indicated more clearly.

Eco-efficiency as a subject is well established but diverse, or rich if we want to emphasize the positive side of this diversity. The richness not only stems from different terminologies developed in different domains involved (Huppes and Ishikawa 2005), but also from more basic, underlying theoretical approaches to this integrative subject. One might avoid the cumbersome details of explicit modeling and evaluation as advocated in this article and go for direct practical solutions, as advocated by Seiler-Hausmann and colleagues (2004) and Bleischwitz and Hennicke (2004). Solutions then relate to management approaches and strategies, using material flow analysis (MFA), and developing sets of sustainability indicators, all depending on the situation. This, of course, is useful and necessary but does not answer the question of what is to be achieved by the management strategies, in terms of economic and environmental goals, and combined as eco-efficiency. We prefer to keep separate the empirical analysis; the evaluation; and the drivers of sustainability, as suggested by Ekins (2005). So, before arriving at solutions, the basic question remains: why eco-efficiency? Does not society have enough environmental quality standards and quality goals, and instruments to realize them, if only enough political will were present? The answer clearly is ‘no’. For policy development, political opinion formation, and well-considered private action, an integrated view, translated into well-defined methods and procedures for weighing economic and environmental aspects, is lacking. Without them, it is difficult to say what is good, not so good, and very good, beyond the simple situation where environmental improvement is possible without cost and without side effects. So we first go into the why question, with answers suggesting that it is important to begin by being more precise on the what, that is, the subject to which eco-efficiency is referring. With the why and the what established, the how questions remain: how to quantify the economic part and the environmental part of the eco-efficiency score, and how to combine these scores into the desired eco-efficiency ratio, where some scaling problems arise. The final framework subject is how to link the analysis to applications, the proof of the pudding. We indicate a few main lines of thought, referring to policies, investment decisions, and product and installation design and development.

1.2 Eco-efficiency for sustainability

Sustainability refers to reconciling environmental, economic and social concerns both from a current point of view and long term intergenerational perspective. Making the jump from concept to tool is loaded with ethical-normative and practical modeling complexities, which cannot be resolved in a broadly acceptable way. Different opinions exist, for example, on the exchanges allowable between the economic and the ecological domain, reflected in positions on (very) strong to (very) weak sustainability (see Neumayer 2003). Eco-efficiency analysis as advocated here does not take a stance on such issues but tries to straighten out the underlying empirical analysis which may show that we are on a path of very strong or of very weak sustainability. To be open to such different options it is essential not to aggregate environmental and economic aspects, but leave them as separate entities as one input into the discussion on strong versus weak sustainability. However, in using eco-efficiency analysis for practical decision support at a micro level of specific firms, products and technologies, and for policies related to these, some link to an encompassing concept of sustainability has to be established, as limited as possible to be open to different positions, but allowing for some broadly agreed upon practical guidance. In simple situations, choices may be clear as when between two options one is superior both in environmental and in economic terms. A simple dominance analysis then suffices. However, in practice such situations are limited and usually some trade-off between economy and environment is involved. Guidance on the trade-off can be given based on broadly accepted assumptions. There is broad support for the position that economic growth should not lead to a deteriorating environmental quality, reflecting a *not-so-strong sustainability* point of view.

1.3 Eco-efficiency, economic growth and Factor X

The simple fact is that quantified eco-efficiency is needed is for analyzing the micro level conditions for simultaneous satisfaction of the rising consumption of a growing global population and attainment of reasonable environmental quality. Spoiling the environment for no good reason seems foolish. But whether we are foolish as a society - or better, how foolish we are - is difficult to tell if there is no method for answering the right questions in this respect. It is not one question that is to be answered, such as “how high is our eco-efficiency?” The real question is how society can support a high standard of living with a high environmental quality, with several questions related, of which a number refer to eco-efficiency, both at a micro and macro level. Discussion of effectiveness of actions, in terms of a certain quality of the environment to be reached at a macro level, and the eco-efficiency of measures at a micro level, which is related to that environmental goal but not in a direct way, has been long-standing. One may take a series of relatively eco-efficient micro-level measures, and even improve their eco-efficiency in time, but never arrive at the desired or required environmental quality; see, for example, the arguments of McDonough and Braungart (2001). Economic growth eats away the improvements per unit of consumption. One may therefore leave the realm of eco-efficiency and - seemingly - pursue effectiveness in a more direct way, as has been done in the past in the Factor X discussion, for example, realizing a Factor Ten improvement in all products in 40 years time (Factor 10 Manifesto, p. 13). This sounds impressive, corresponding to an improvement in environmental impact per unit of product of 6% per year. However, a rise in consumption by 4% per year may reduce the Factor Ten effect substantially, leaving only a Factor Two in 40 years time. If, in this same period, rising affluence leads to consumption shifts in an environmentally more stressing directions, for example, more traveling, more meat consumption, and more air conditioning, the net environmental effectiveness may well be negative despite realizing Factor Ten in all products. The effectiveness-related objections to eco-efficiency seem to miss the point that it is not the concept that is wrong but the eco-efficiency improvements at a micro level that are quantitatively insufficient for reaching the environmental quality goal at a macro level. Trying to link the micro level directly to the macro level seems an inappropriate route; macro level developments play their independent role. Because the eco-efficiency concept can be applied at the macro-level as well, to regions and countries, some of the discussion of effectiveness is part of the eco-efficiency analysis, such as analyzing the eco-efficiency of recycling in a region (see Morioka et al. 2005 and Seppälä et al. 2005).

So let us get back to the ultimate problem faced by society: Economic growth is increasing in large parts of the world and environmental assets are fading fast, globally. The Millennium Ecosystem Assessment (2005) has shown that a sad deterioration of all major environmental assets was taking place in the world even before economic growth in China and India picked up to current high levels. In many instances the scientists involved see no good reason at present for a reversal of such downward trends, because the drivers of these developments are the aspirations of all global citizens to become affluent, and firms and governments set a high priority on accommodating those wishes. Even the richest Western societies strive for more and more, because there is no real limit to demand, and working more may enhance international competitiveness. Without taking this desire for granted, it must be accepted that some level of growth will be present for a long time to come, not only driven by demand but also by R&D and resulting technological advances on the supply side. If economic growth cannot be redirected substantially into an environmentally benign direction, the trends of the Millennium Assessment are unavoidable. Next, why are the simple tools of the past not applicable to the future? We have more or less solved the ozone layer depletion problem and, in the Western world, the acidification problem; why not tackle all problems like that? The answer is that in many instances, the simple banning of substances, or a limited number of end-of-pipe measures, is not an option any more. Such low hanging fruit has mainly been picked, whereas the sheer size and complexity of economic activities have risen to unprecedented levels, as part of globalization. Somehow, our globally connected actions at a micro level have to be reconciled with environmental quality at a macro level, most actions directly and indirectly having consequences for most environmental problems on earth. Therefore, the multifaceted environmental quality goal must be translated back to the level of decision making at a micro level, be it for public policies or for enlightened actions by firms and individuals. It is a requirement for such policies and actions to reconcile market-related economic welfare with the environment. One cannot hope to grasp this all in two numbers, but the nominator and denominator of eco-efficiency clearly are of central importance.

New environmental problems, such as ocean fisheries depletion, and problems not so easily linked to specific economic activities, such as global species mix, do not fit well into the eco-efficiency framework. Even if only the emissions of hazardous substances, including eutrophicating substances, could systematically be brought into the analysis, though, the simplification in decision making would be enormous, freeing regulatory power and intellectual capacity for solving other, less directly linked environmental problems, and also for solving social problems. Eco-efficiency is not only relevant for general cost considerations. At a political level, the power of the market and the urge for full employment are very strong. If, for fear of the crudeness of simplification of analysis, eco-efficiency is not defined and established, it is not so much the reduction of affluence that will result due to inefficiency, but a less effective policy and lower environmental quality. Reducing the cost of environmental improvement by increased eco-efficiency thus is a means to higher environmental quality as well. So answering the why question: leads to the what question: the eco-efficiency of what should be improved to shift society toward higher environmental quality?

1.4 Definitions of eco-efficiency

A wide variety of terminology referring to eco-efficiency has been developing, differing depending on application, on the background of the researchers, and possibly even on views on how to treat negative signs. Some autonomous divergence is also present, because subgroups involved in the discourse do not refer to each other. As a result, the term *eco-efficiency* is used in different ways and other terms are used that overlap with these meanings, such as environmental cost-effectiveness and environmental productivity. We try to bring some order into this usage, distinguishing between the formal definition and the specific content given to the variables involved. We focus on the formal definition here. The content given to cost and value, as economic categories, has been widely standardized in accounting conventions—see the publications lists of International Standards of Accounting and Reporting (ISAR 2005)—and ideally fits into the framework of national accounting as actively standardized under United Nations coordination in the System of National Accounts (SNA 2002). In analyzing the eco-efficiency of a new technology or product, however, aggregate accounting frameworks may miss essential detail and related effect mechanisms. Hence, they cannot be the last word. Management-oriented concepts such as cost-benefit analysis (see, for example, Mishan 1971 and Dasgupta and Pearce 1972, both for public applications) and life-cycle costing (see, for example, Fisher 1971 for public applications and Dhillon 1989 for private applications) may then be more appropriate for public and private applications but also lack standardization. For the environmental part, no such detailed standards exist. A great variety of theoretical and practical approaches have emerged, in parallel at best, but often overlapping. The standards for life-cycle assessment, ISO 14042, developed by the International Standards Organizations, give only a few guidelines. Work by the Society for Environmental Toxicology and Chemistry (SETAC), now incorporated into the Life-Cycle Initiative of the United Nations Environment Program and SETAC, is more detailed but has not yet led to broad acceptance of specific methods. Though it is of prime importance for the eco-efficiency discussion, we will not venture into this subject here. Here, we assume a normal, albeit complex, situation in which environmental aspects of decisions cannot be encompassed by just a single environmental intervention, such as emission of carbon dioxide (CO_2) or sulfur oxides (SO_x), but relate to a usually large group of environmental interventions. These in turn link to the environmental effects mechanisms that follow interventions, such as climate change, acidification, and summer smog formation, which in turn relate to areas of protection such as human health, ecological health, and human welfare. So more encompassing concepts are needed to represent the environmental part of eco-efficiency, which have not yet been filled in a comprehensive and broadly accepted way. Contrary to specific applications meant for eco-efficiency, as in the business orientation of the World Business Council for Sustainable Development (WBCSD), the concepts defined here are generally applicable to choices regarding both production and consumption and to choices regarding public policies and private choices, both of a practical and a strategic nature.

Eco-efficiency has been defined as a general goal of creating value while decreasing environmental impact. Leaving out the normative part of this concept, the empirical part refers to a ratio between environmental impact and economic cost or value. Two basic choices must be made in defining practical eco-efficiency: which variable is in the denominator and which is in the numerator; and whether to specify environmental impact or improvement and value created or cost. Distinguishing between two situations, the general one of value creation and the specific one of environmental improvement efforts, and leaving the numerator - denominator choice to the user, as diverging practices have developed, four basic types of eco-efficiency result: environmental intensity and environmental productivity in the realm of value creation; and environmental improvement cost and environmental cost-effectiveness in the realm of environmental improvement measures.

1.5 Choices in terminology

The starting point for the formal definition of eco-efficiency is the general definition of WBCSD (1992, 2001; an overview can be found in DeSimone and Popoff 2000), which goes back to the work of Schaltegger and Sturm (1989). They describe eco-efficiency as a ratio between two elements: environmental impact, to be reduced, and value of production, to be increased. We disregard the normative overtones again, looking at eco-efficiency as a measuring rod only. The value of production lies in the products produced, comprising both goods and services. Two equivalent variants are used, the ratio of value to environmental impact (for example WBCSD 2001) and the ratio of environmental impact to value (for example UN 2003), one being the exact inverse of the other, but with the same information content. In addition to the creation of maximum value with minimum environmental impact, there is the analysis of dedicated environmental improvements (see for example Hellweg et al. 2005). The focus then shifts from the creation of value to the reduction of cost for the environmental improvements investigated. The signs of both numerator and denominator then reverse, or the variables are defined in the opposite direction. This distinction between the analysis of value creation and the analysis of environmental improvements can be combined with the inversion options. It seems wisest to make eco-efficiency an overarching general concept, with variants residing under this umbrella.

Table 1.1 Four basic variants of eco-efficiency

	Product or production prime	*Environmental improvement prime*
Economy divided by environment	Production/consumption value per unit of environmental impact: **1 environmental productivity**	Cost per unit of environmental improvement: **3 environmental improvement cost**
Environment divided by economy	Environmental impact per unit of prodution/consumption value or: **2 environmental intensity**	Environmental improvement per unit of cost: **4 environmental cost-effectiveness**

The relationship of these variants is shown in Table 1.1. In actual applications, there often is not a full system being analyzed but a difference analysis between options is performed, with positive and negative results depending on which situation is taken as a reference. For example, in a win-win situation resulting from technological improvement, described as a difference from the current - or not improved future - situation, the denominator of environmental productivity becomes negative, as then does the ratio itself. Similarly, some environmental improvements may not entail cost but reduce cost as, for example, by creating additional value. Then the environmental cost-effectiveness becomes negative. Making separate categories also for these cases would lead to a confusingly large number of terms, because, for each of the four basic options, the sign of the numerator, of the denominator, or of both may change. If all these situations were really distinguished, 16 options would result. The reason for discerning them is that the principle of “higher (or lower) is better” does not hold any longer with a sign change, nor when absolute values are taken. It seems better to treat such situations in a practical way on a case-by-case basis. Such special cases may easily be subsumed under any of the four basic variants of eco-efficiency. Along with these four basic eco-efficiency terms and concepts, there are similar concepts, with related meanings, such as energy productivity, (primary or total) resource productivity, capital productivity, and labor productivity, with each one having the corresponding intensity as an inverse, see Heijungs (2006) in this book. As he describes, a group of terms relates to technology discourse, where there is an input-output efficiency referring to the same variable occurring both as an input and as an output, with efficiency being the complement of the loss factor. Examples are resource efficiency in kilograms/kilogram and energy efficiency, in joules/joule. The eco-efficiency terms, alas, are not in line with this technology-oriented terminology. In eco-efficiency, the environmental impacts and the economic impacts both relate mainly to outputs of the activities involved in production, consumption, and disposal management. Of course, such input-output concepts might be subsumed under the eco-efficiency umbrella, leading to additional types.

The basic terminology proposed here deviates slightly from the one used in most eco-efficiency publications, by being more encompassing and by having two levels of generality. It has the advantage that it clarifies formal meaning, while leaving specific content open to a next level of more detailed discussion. This terminology proposal is meant for easier communication. Of course a consensus on terminology requires a broader social endeavor, involving the many fora involved. Organizing the consensus-formation process is hampered by the decentralized nature of the eco-efficiency community. A cross-cutting organization such as the temporary eco-efficiency conference community resulting from focused conferences might form a most practical path.

So, summarizing, we distinguish four main types of eco-efficiency (Huppes and Ishikawa 2005). The first two are environmental productivity and its inverse, environmental intensity of production, referring to the realm of production. The second pair, environmental improvement cost and its inverse, environmental cost-effectiveness, are defined from an environmental improvement measures point of view.

1.6 Eco-efficiency of what?

Eco-efficiency, as a ratio between economic value and environmental impact, may be applied to any unit comprising economic activities, as these activities always relate to cost and value, and having some physical substrate, always influence the environment. The units may encompassing, as comprising total society, that is the macro level. Several options exist for units at a more micro level of aggregation, as involving sectors, technologies, product systems, regions and countries. We treat the micro and macro options in this order. So we first specify the eco-efficiency of products and technologies at a micro level, as in Figure 1.1, and then sketch the relation to eco-efficiency at the macro level, as in Figure 1.2. Some more detail on these micro-macro relations, such as between GDP, factor incomes, and costs of firms, may be added (Kuosmanen 2005), but is not required yet in this framework analysis. Also, the relation between value and capital is not explored, though clearly relevant in the context of intergenerational sustainability analysis, to which eco-efficiency should contribute. Figge and Hahn (2005) explore this subject and indicate that, starting at the level of economic, environmental, and social capital, the eco-efficiency of firms may be defined. Finally, we make some remarks on environmental effectiveness in a dynamic context, which is more realistic but also more complex to analyze, with elements such as sunk cost, technological lock-ins, saddle points secondary effects of decisions as in income effects and rebound effects, macroeconomic mechanisms, and political limitations. The conflict between realism and easy practicality is a central subject there.

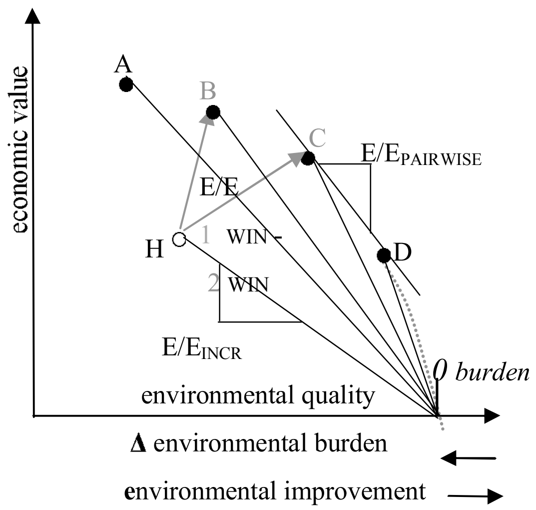

Figure 1.1 Eco-efficiency of technologies: E/E_{INCR}, $E/E_{WIN-WIN}$ and $E/E_{PAIRWISE}$. H represents a current historical reference situation and A to D are new technical options

Micro-level eco-efficiency of technologies

In Figure 1.1 we depict three basic options for applying eco-efficiency at a micro level, each with its own numerical outcomes. This figure assumes a given amount of production factors being available, with each point indicating a production possibility for society. Of course in practice a firm may opt for simplification of the optimality requirements related to the selection of production factors for application in specific activities and technologies. The first application, incremental eco-efficiency, E/E_{INCR}, specifies the effects of the total value of a product system or sector and its total concomitant environmental effects, for example, as environmental productivity. It is depicted for a number of technologies by the lines starting from zero burden. One may further differentiate within these totals by indicating the effects of one incremental unit of production. This difference of course shows only if the model can specify (dis-)economies of scale. The curved dotted line OD depicts the marginal eco-efficiency of one unit of production. As it is not short term optimization in which we are interested, the incremental technology unit E/E_{INCR} may be interpreted as long term marginal analysis, adapting all capital goods to the intended volume of production or consumption. This nonlinear analysis is hardly ever available in simple practical economy-environment models such as life-cycle assessment (LCA)-based models, which are linear homogeneous. In such models, the average score and the one-unit-incremental score are identical. For models with results depending on scale, such a shift at the boundary is very similar to comparisons between technologies A, B, and so forth; see below. To avoid a further terminological differentiation, we do not treat this as a separate option. We refer to the full market volumes here as incremental eco-efficiency and reserve the term marginal eco-efficiency for comparison between technologies; see below. The second application is $E/E_{WIN\text{-}WIN}$, which gives a comparison between a historical reference situation H and potentially new situations based on the use of improved technologies, here A to D. Options B and C then depict win-win situations. Of course if a still more inferior historical reference is chosen, to the South-West of H, more situations fall under the heading of win-win. The Factor X analysis also falls into this category, but then is not based on the monetary value of a product but on its physical units with a certain utility, as in LCA with its functional unit. The disadvantage of having an irrelevant alternative (if the old option is obsolete) as a reference is that all numerical outcomes depend on it, and hence also all eco-efficiency scores. Using such a measure in broader optimality analysis goes against the basic rule in social choice theory of independence of irrelevant alternatives (Arrow, 1970; Sen, 1970). However, it is quite handy in indicating the amount and rate of progress in specific technology development.

The usefulness of such win-win analysis hence is limited, because it cannot give guidance on the question of whether the win-win realized is good enough for society to adequately improve its overall environmental performance. For example, it may well be that win-win situation B involves an economy-environment trade-off that would be highly destructive of the environment if applied throughout society, because it leads to environmental burdens ten times higher than option D. An example might be in energy production, where shifting from coal-fired power stations, as option H, to integrated gasification combined cycle power production might constitute option B. Large scale carbon sequestration is, however, needed to reduce climate-changing emissions to desirable levels, represented by option D. The third micro-level eco-efficiency application, difference eco-efficiency or $E/E_{PAIRWISE}$, is similar to the win-win variant, as also here two alternatives are compared. But its use is totally different. First, it is applied to remove all irrelevant alternatives, that is, those lying within the concave envelope created by the most attractive options. Option H, being dominated by B and C in decision-theoretical terms, does not belong to the potentially optimum set of technologies and hence is irrelevant in decision making. When all such irrelevant alternatives have been removed, the envelope of potentially optimal technologies remains. The difference analysis between two adjoining technologies on the optimum envelope may, ideally, be transformed into a marginal analysis, indicating the trade-off at the point of that technology implied by a shift in the one or the other direction. We use the term marginal eco-efficiency for trade-offs at this optimum envelope, both for specific technology domains and for society at large. Which of the technology alternatives is actually optimal depends on how we see the trade-off between economic value (in constant prices) and environmental value, from a normative point of view. If we put a relatively low weight on environmental quality, option A becomes best, but with a high value on environmental quality option D is to be preferred, with B and C falling in between. For orthodox neo-classical economists, the units on both axes n principle are the same: utility as represented by its monetary value. Then, after such scaling, the trade-off is given as 1:1. If the axes are not in the same unit, the value choice of relative importance of economy vis-à-vis environment is to be made explicitly in order to define what is optimal, linking the two different variables involved. This is the general situation for non-orthodox economists as well. Ultimately, with all normative trade-offs defined, the non-economist and economist approaches do not differ so much, because with appropriate rescaling of the environmental axis, the trade-off per unit can be arranged to become 1:1. The basis for integration of environmental aspects into a single score may be very different, however, giving a different meaning to the outcomes. One interesting consequence of this trade-off analysis is that the difference in application to full production volumes (as in the approach to eco-efficiency used by the World Business Council on Sustainable Development - WBCSD2000) versus eco-efficiency analysis of specific environmental improvement measures (as in contributions by Scholz and Wiek (2005) and Hellweg and colleagues, 2005), as environmental cost-effectiveness, vanishes. They both are marginal eco-efficiency analysis (in the terminology used here), to be evaluated in the same marginal eco-efficiency framework as used for process- integrated alternatives. A further consequence of this analysis is that the link to macro-level analysis now can be specified in a way that connects to optimality analysis for society, which ultimately forms the broadest level of justification for eco-efficiency analysis.

Macro-level eco-efficiency of society

The ultimate aim of eco-efficiency analysis is to help move micro-level decision making into macro-level optimality. This in turn is based on the environmental quality society seeks, given a specific level of economic development, as macro-level eco-efficiency to be attained. The trade-off society makes normatively determines what is optimal, as one point on the societal production possibility curve. So the sum total of all production factors corresponds to a set of potentially optimal points on the production possibility curve (see, in this vein of thought, the work of Bator 1957). Potentially optimal points from the domain envelope, each with the same trade-off point, add up to a point at the societal envelope having that same trade-off. If in society in one domain a certain trade-off is realized and in another domain a different one, they add up to a point within the societal envelope of potentially optimal points. Hence such a point cannot itself be optimal. This leads to the for some people counterintuitive consequence: improving the environment by increasing the trade-off in a certain domain to the right downward part of the envelope, for example, building quite environmentally friendly but extremely expensive solar power installations may detract from absolute environmental quality, because welfare losses of a smaller amount could have realized larger environmental gains. Of course experimental application may be a useful part of product development, as an R&D effort.

When linking to this macro-level analysis, we assume that different studies on eco-efficiency use the same units for economic value or cost and for environmental impacts and benefits. In actual applications, this hardly ever is the case. Even if using the same impact categories, in many studies the two axes are normalized relative to some alternative or to an average of some set of alternatives, whereas others normalize only the environmental axis in such a case dependent way (Kobayashi et al. 2005; Rüdenauer et al. 2005; Suh et al. 2005). Eco-efficiency scores, seemingly comparable, then are not due to differences in scaling on the two axes. Placing the eco-efficiency analysis in a broader societal efficiency context requires a case independent specification of the axes (see work by Heijungs et al. 1992 and Norris 2001). Without having the same units on the axes for different cases, no comparable trade-offs can be quantified and an inter-case analysis becomes impossible. Applicability of eco-efficiency then reduces to specific domains of application. This still is useful for eliminating suboptimal variants at that case level but does not fit into the macro level analysis we think ultimately is required. But given the conceptual problems involved in specifying a normatively valid trade-off between environmental aspects, one can hardly expect results to have high validity at a case level now. For linking the micro level to the macro level, the starting point for adding economic and environmental effects of all technologies is a hypothetical zero-burden situation; see Figure 1.2. By simple addition, total environmental burdens of all technologies together may be related to total environmental quality, as E/E_{TOTAL}. Let us first start with an actual situation, which is such a sum-total of all actual economic activities in society. Technologies in society are added, starting from the zero-burden point, until the total production & consumption volume is covered. The lines depicting technologies indicate their contribution to economic value and environmental burden, as incremental eco-efficiency. Their total depicts the similar measure for society, as environmental intensity, or the equivalent inverse, environmental productivity. In macro-level studies, such as of decoupling of economic growth and environmental quality, environmental intensity is customary, defining eco-efficiency, for example, as environmental impact per unit of national income.

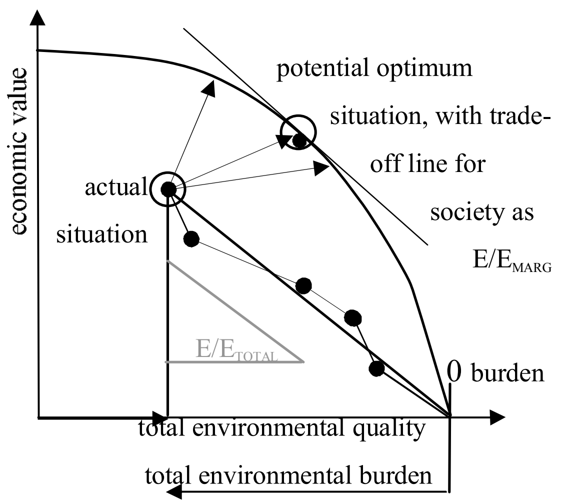

Figure 1.2 Eco-efficiency in society: Actual technologies, E/E_{TOTAL}, and production-possibility envelope with trade-off line E/E_{MARG} for society. Black dots denote specific technology domains

The marginal eco-efficiency score, based on pairwise eco-efficiency scores for each (black dot) technology domain relative to next possible options, has not been indicated for each technology domain in this macro-level figure. It could be depicted as a small, also concave curve within the envelope, bordering on the societal envelope. A rigorous mathematical treatment of this now graphically treated subject is still lacking. Without an explicit goal for the relevant trade-offs, we can be sure that this optimality score will be different per technology domain in practice. As a consequence, the actual situation facing society will not lie at the envelope curve of potentially optimal situations, where each point assumes a systematic choice based on the same trade-off for all choices in all technology domains. By indicating the distance from the actual situation to a point on the envelope, the "avoidable" sub-optimality is indicated, always relative to a normative choice on the economy environment trade-off. As discussed above, contrary to common intuition, both technologies with a higher value for the environment in their trade-off and those with a lower value contribute to the sub optimality. Each level of normative trade-off defines a point at the envelope, which then links to choices at the micro level with the same trade-off for all technologies in society. Clearly, our actual situation is not at a potential optimum point on the envelope. Making decisions in the right direction thus is not a straightforward affair. Should we focus on slow but fundamental improvements or is a catch-as-catch-can strategy the better option? Thus in real life more aspects must be taken into consideration than those of eco-efficiency itself. Should we accept that shifting investments between sectors is not possible? In a second-best world it may be wise to accept different trade-offs in different technology domains or sectors, for the time being, and actively search for less sub-optimal solution in the longer term.

Dynamic eco-efficiency

In reality, the technologies set as constituting the efficiency boundary assumed above does not exist or, more precisely, it is not well defined. We cannot shift between technologies at will, because such shifts involve adjustments in society, in the volume and nature of the capital goods industry, in terms of transport infrastructure, adaptations in regulations (not only environmental ones), and so forth. Also, at any point in time, in each technology domain, new technologies are emerging that lead to different sets of optimal technologies and hence to changing marginal eco-efficiency for the technologies considered. In reality, all technologies are path dependent, see for an early advocate of this non-classical approach Schumpeter (1943). All optimal technologies will become suboptimal in the course of technological progress. Implementation of new optimal technologies not only requires time but, even if possible, should not be done too fast. Installing any new technology directly would imply a continuous destruction of installed capacity, even if the destruction is creative also creating environmental costs. So adding real life dynamics would make the analysis much more meaningful, and much more complex. In shifting to full causal analysis—as is the essence of dynamic modeling—the easy aggregation by addition of technology domains has to be replaced by a causal model, indicating effects of choices and actions. Ideally, such an analysis indicates how the future would be different as a consequence of the choice made. Because this involves predicting the future twice when analyzing just two alternatives, this is a most demanding approach. Requiring real dynamic modeling for decision support would make eco-efficiency analysis, and any optimality analysis at the level of technologies, practically impossible. When eco-efficiency analysis is applied to practical decision making, the limitations of non dynamic analysis should of course be considered, at least in a further qualitative additional analysis. For now, more modest aims may be set for the analysis, starting with the simpler comparative static analysis depicted in Figure 1.3. This may be a starting point for deepening the analysis: how to deal with sustainability, including social aspects; how to consistently reckon with spatial and temporal aspects; how to relate to practical decisions in an appropriate way; and not just having solutions but making them consistent and transparent (Brattebø 2005). This simplified addition of dynamically relevant aspects is manageable in practice. Such simplifications are the more important because eco-efficiency modeling should be broadly applicable, including applications to decisions by consumers and by small and medium sized enterprises (SMEs) (Suh et al. 2005). Acknowledging the limitations of comparative static analysis, some insights may still be gained. First, the level of the trade-off, however disputed it may be normatively and politically, can be seen as an actual characteristic of society as exhibited in choices on technologies and policies. Different choices on marginal eco-efficiency in different domains clearly are a sign of sub optimality, assuming they do not result from deep dynamic insights. In developing new technologies, such indicative trade-off relations (Oka et al. in this book and 2005; Kuosmanen 2005) may roughly guide choices, leading to the development of a relevant domain of new technologies with substantially higher eco-efficiency.

Also, one may assume that with rising incomes in the course of time, the normative trade-off will shift toward more emphasis on environmental quality. Poor people cannot afford high costs for environmental improvement. This reasonable but not proven assumption may also guide choices in technology development in the right direction, that is, toward a range of feasible future trade-offs between economy and environment. In each domain, technology development will have to take place, leading to an envelope

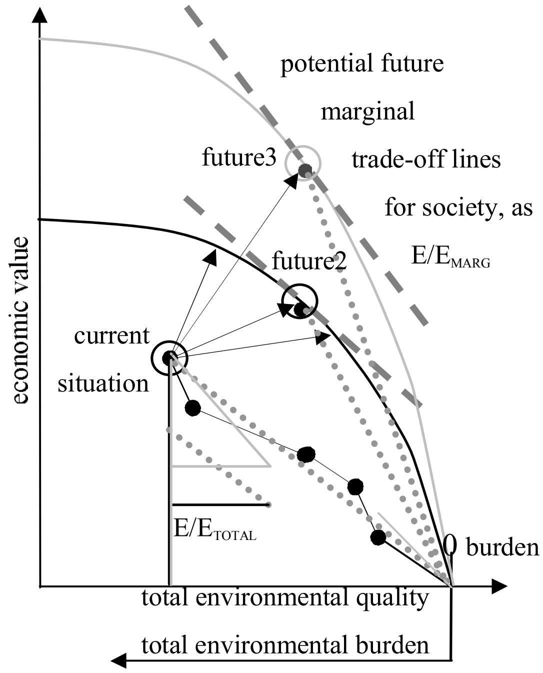

Figure 1.3 Dynamic eco-efficiency in society: shifting trade-off lines

of non dominated alternatives with a steeper curve from the zero burden origin. For society as a whole, this means that the environmental intensity curve will move upward, with the zero burden point remaining fixed. What can we learn from this theoretical dynamic exercise? At first sight, results are not comforting. All actual technologies improve as we grow, leading to the gray lines (with dots) in Figure 1.3 all becoming steeper because new technologies have been implemented. Damage per money unit of consumption decreases, but total damage remains constant. We may all become more affluent, but the environment will not improve. Only if we assume that actual trade-offs will shift, with more emphasis on the environment, may we move to a point where environmental burdens may decrease absolutely. If actual technologies lag behind optimal ones in the same proportion as they do now, this reasoning also holds for the suboptimal state we are in and will be in. Our efforts, already substantial, to remain constant in environmental burdens will have to go well beyond this for actual environmental quality improvement while we grow, or the warning of one of the first environmental economists will come true: "as ye grow so shall ye weep" (Mishan 1969, cover).

To avoid environmental regret on economic growth, two steps are essential both involving eco-efficiency analysis for their practical application. The first is to help move society in the direction of optimality, avoiding both too environmentally costly value creation and too high cost for environmental improvement. This is moving from current situation 1 to the more optimal situation 2 in Figure 1.3. The next step is to help guide economic growth. If economic growth takes place with the eco-efficiency of activities remaining the same, the environment will deteriorate. Even very weak sustainability requires eco-efficiency to move into a more environmentally benign direction, that is the steeper striped line in Figure 1.3.

1.7 Economic score

In the process of arriving at eco-efficiency ratios, the market part is to be quantified in one term, as cost or value, and the environmental impacts are to be aggregated into one score as well. Value and cost aggregation are well established subjects in two main domains, cost-benefit analysis (CBA) and life-cycle costing (LCC), both developed in the middle of the 20th century. Cost-benefit analysis has a broad societal point of view, disregarding transfer payments and correcting market values for market imperfections (for classics on this topic, see Mishan 1971, and Dasgupta and Pearce 1972). Like LCC, it takes a full systems point of view, covering "the life cycle." Life-cycle costing, as developed for public procurement by the Rand Corporation in the United States, see for example the work of Fisher (1971), and by management accountants for application in firms, see for example the work of Dhillon (1989) both take a budget point of view, including transfer payments such as taxes and subsidies, and accepting the actual functioning of markets, including capital markets. Though for each approach different aggregates are possible, for example, as related to value-added or cost concepts, the underlying reasoning is well established and will not be much discussed in this volume. Both CBA and budget related LCC can express cost or value as a discounted present value. In the realm of LCA, discussions on how to align cost accounting to steady state LCA modeling, directly related to the eco-efficiency subject, may give rise to steady state cost or value as a third approach to LCC (see work by Rebitzer and Seuring 2003 on the LCA related SETAC Working Group on LCC and the survey by Huppes and colleagues 2004). Some conventions on specifying cost and value might come in handy, though, at least in specifying which approach is followed, how empirical effects are modeled, and which aggregation method is applied. For example, when eco-efficiency is analyzed from a broad societal perspective, as in analyzing climate-change policy measures, the logic would indicate a CBA type of cost and value analysis, such as the Intergovernmental Panel on Climate Change (IPCC) does in its publications (IPCC 2001). In CBA, though, economists tend to express market value and external effects as referring to the same value concept. This final integration step of external effects with market related magnitudes may better be postponed and, if done, be made as a recognizable last step, for several reasons. These reasons relate to, for example, the uncertain nature of environmental effects; the impossibility of specifying all effects in terms amenable to subjective evaluation by consumers; the lack of agreement on discounting when long time horizons are involved; the Brundtland principles of intragenerational and intergenerational justice and equity; and the divergence in stringency of actual environmental policies. So, in CBA for eco-efficiency analysis, the environmental external effects are kept distinguishable from market-related effects, avoiding at least some of these issues of contention.

In budget LCC and LCA-related LCC, cost and value refer to market related items only. For a given cost and value concept, numerous empirical issues must be resolved, especially if long time horizons are involved. In their comparative study on eco-efficiency trends, Dahlström and Ekins (2005) encounter the problem of changing market values of steel and aluminum, directly influencing the eco-efficiency scores. Historical studies may solve such issues by giving time series of prices as well. For future oriented studies for decision support, historical values are proxies for expected future prices. Especially for abiotic resources, which have shown substantial long term price decreases and volatility, expected prices may be highly disputed, and hence the eco-efficiency of decisions involving such resources as well. Uncertainties concerning the future cannot be avoided, but may be made visible to some extent by scenario development on main uncertainties. These then are reflected in ranges of eco-effiency scores, as a certain softness in results.

1.8 Environmental score

Environmental effects are those resulting from the choice at hand. Economic activities jointly produce environmental effects, for fundamental reasons, both in terms of resource extraction required for production, the environmental inputs, and in terms of losses from production, consumption, and waste management, as outputs to the environment. These relate to the first and second law of thermodynamics (see for example Baumgärtner et al. 2001 for a survey). Ultimately, there is no free lunch in environmental terms. But the environmental effects of all our lunches are far greater than thermodynamically determined minima, as calculated in terms of energy and exergy analysis (Baumgärtner and de Swaan Arons, 2003). Such analysis does not link in any direct way to biodiversity effects of economic activities. Clearing tropical rain forests is not a matter of thermodynamics but of socioeconomic and political dynamics. Though thermodynamics unavoidably rules, the choices we have go far beyond these physical constraints, and our environmental concerns, such as those in terms of human health and ecosystem health, cannot be reduced to thermodynamics analysis alone. So a main subject of diverging opinion in eco-efficiency analysis, not yet based on firm analytics, is how to specify and aggregate environmental effects. The intention is to cover all relevant environmental information, as the empirical part, and aggregate these empirical effects in a way that leads to a broadly acceptable single-score result, either focusing at efficiency only, as in Maximum Abatement Cost (MAC) method, or at least partly based on value judgments or preferences. With the relevant variables defined and agreed upon, the empirical part of effect (or: impact) analysis again is fraught with traditional problems in decision theory, with subjects such as before-and-after, with-and-without, indirect effects of varying complexity, and conditionality on compensating measures. Again, these subjects deserve attention and at least a specification of actual choices made in these respects. Which environmental effects are to be specified of course remains open to discussion. The United Nations propagates one specific method for impact assessment in the context of eco-efficiency reporting (UN 2003). In the realm of LCA, a survey of methods for environmental impact a nalysis is provided by Guinée and colleagues (2002) and Udo de Haes and colleagues (2002). These methods often originate in the public domain as efforts to standardize environmental analysis as part of the policy process such as, for example, in Japan (Itsubo and Inaba 2003), in the United States with the software Building for Environmental and Economic Sustainability (BEES 3.0 2004), and in the Netherlands (Guinée and colleagues 2002). Steps toward international standardization are ongoing, such as in the United Nations Environment Program—Society for Environmental Toxicology and Chemistry (UNEP-SETAC 2005) Life-Cycle Initiative. One point of basic agreement is on the distinction between *environmental interventions*, such as emissions, extractions, and land use; their *midpoint impacts* through main environmental mechanisms such as global warming, acidification, and toxicity; and the *endpoint impacts* of ultimately relevant items as related to human health (e.g., as morbidity and mortality), to environmental quality as an independent value and as the life support system (e.g., as biodiversity), and to human affluence (e.g., as reflected in production functions, landscape, and cultural heritage). Again the broad discussions going on in this field should be acknowledged when specific choices are made, but we will not go into them here. The focus here is on how environmental effects, when specified somehow, may be aggregated in a more or less generally accepted way. This acceptance is based on reference to what others in society have as views, values, or preferences. Two basic dimensions may help survey the field and clarify actual approaches. One is whose views and preferences are represented; the other is how they are expressed (see Figure 1.4). Whose views and preferences is it that have a general acceptance? In one approach it is all citizens in society, which is the economists' approach, or the direct democracy approach. Somehow individual preferences on environmental effects are aggregated into a social welfare judgment; see Arrow (1970) and Sen (1970) as main contributors to the analysis of this field.

In another, less formalized approach, the aggregation is through the political process, with public policy outcomes as the basis for the aggregated view. For both approaches, a fundamental problem is how to know the private and public preferences, either with stated views as a basis or with preferences derived from actual choices. In economics, broadly applied methods are interviews and panel procedures to measure the

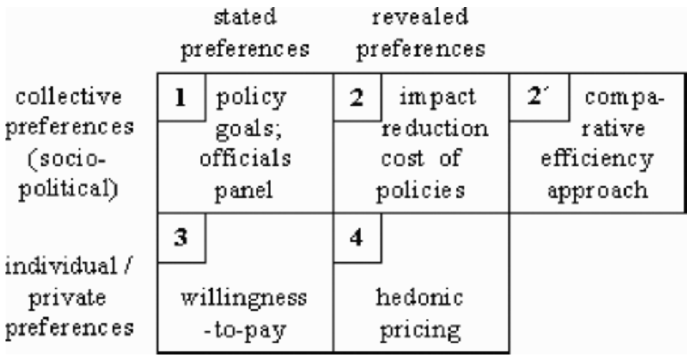

Figure 1.4 Five main types of aggregation for eco-efficiency analysis

willingness-to-pay for avoiding environmental effects (or to be paid for accepting them). The other option is to derive the preferences from actual choices, such as hedonic pricing, as for example inferred from lower housing prices for similar houses in more contaminated areas. Collective preferences similarly can be derived from public statements, as in policy goals in policy documents, or through interviews and panels with public officials. Or they may be derived from actually implemented policies, reflected in the cost deemed acceptable for their implementation, as revealed collective preference. Combining the two dimensions, four base approaches result, which can be expressed as weights on environmental impacts or the emissions and other interventions creating them. We will treat them in turn. Of course, it is free to anybody, or to groups of stakeholders in some decision procedure, to create their own weights, presenting their own preferences, or their views on future societal preferences. Such weights do not have the generality and authority striven for in the approaches now discussed in more detail. Special mention is made for an approach related to the revealed preference approach, but avoiding the welfare theoretical interpretation. It focuses on the actual cost of combined emissions reduction stating how efficiency in environmental improvement can be created, without knowing public or private preferences. This comparative efficiency approach is empirically filled in in this book in the paper by Oka and colleagues as the Maximum Abatement Cost (MAC) method.

Stated collective preference

Stated preferences may be derived from stated policy goals and from direct weight setting, as in panel procedures by public officials. In setting policy goals, such as reduction percentages or quality levels to be attained in a certain year, the preferences may be seen as a distance to target. Such distances, though, may already reflect assumed cost, because one would not set goals higher than implies a reasonable cost for reaching them. So, by estimating the expected cost for attaining the goal, a measure of the relative importance of the goal can be derived, in monetary units. An example with practical data on the Netherlands is provided by Davidson and colleagues (2005). Disadvantages of this approach relate to the somewhat ambivalent nature of policy goals, in that stated intention and effective later realization may not match, as seems to be the case in many countries regarding implementation of the Kyoto Protocol obligations. Also, using hypothetical costs of hypothetical technical options to reach the goals may grossly overestimate the more reasonable but vaguely expected "real" cost. Panels with public officials are another option for deriving stated preferences. Only a few examples exist, one from the United States in environmental analysis of building in the BEES software (BEES 3.0 2004), without a clear background of reasoning toward the weighting set being used (see Lippiat and Boyles 2001), and one from the Netherlands in an environmental covenant with the oil and gas industry (Huppes et al. 1997, with an update of the panel results in 2002; see Huppes et al. 2006 forthcoming). The advantage of this procedure is that the weights given can be directly related to specialized knowledge, such as knowledge of impact assessment models and detailed knowledge of the problem mechanisms involved. Application of these weighting sets is specific to a quantified problem description at a normalized level, for the United States, for the Netherlands, as in the case examples, or for other countries, or for the world. The broader application of such weighting sets should be based on more explicit public support for them, which now is lacking.

Revealed collective preference

Using the actual costs of emission reduction or environmental quality improvement avoids the vagueness of intentions and hypothetical technologies.

Especially if cases can be found stating the expected cost of actually implemented environmental measures, good insight into actually used trade-offs can be gained. But things are never so simple. In actually implemented technical measures for emission reduction, the costs are hardly ever assessed for a single emission or a single environmental problem. Reducing sulfur oxides (SO_x) emissions from electricity production by 1 kilogram typically requires 12 kg of carbon dioxide (CO_2) emissions, and has a further influence on virtually all other emissions and resource use in the world. Also, cost may be highly location dependent, as in exceptionally densely populated areas, where general ambient quality requirements are not met. Such incidental high costs cannot be seen as representing overall collective preferences. Wisely selected cases and sophisticated estimation procedures may reduce these problems to reasonable proportions. However, both stated and revealed collective preferences will now lead to diverging results, with consequences for the eco-efficiency analysis in these cases (see Nieuwlaar et al. 2005). Adding individual preferences or values based weighting sets does not solve this problem, to the contrary.

Comparative efficiency

In applying aggregation methods, one might wish to reduce the assumptions being made to a minimum. The most robust system available then is a variant of the revealed collective preference method. Its application may even avoid the interpretation as collective preference, by only stating the relative efficiency of options, relative to a base case, as in the maximum abatement cost method (see this book and Oka et al 2005). This may be seen as a special and practical case of the more general efficiency frontier approach (Kuosmanen and Kortelainen 2005). In applying the resulting weights to cases, one may see if the environmental improvement accomplished at additional cost (or lower value creation) might have been created for a lower price elsewhere. This does not entail any assumption on rational public preferences, just a reference to costs of emission reduction at other places. If enough data are available, cases with multiple emissions can also be covered in this way. This subject is also contentious, because surveys of cost per human life saved (for example as Disability Adjusted Life Years, DALYs) by different measures show widely diverging ranges;

see the survey by the U.K. Department for Environment, Food and Rural Affairs (DEFRA 2004).

All difficulties in the modeling of both costs and environmental effects are present in the cases covered in such survey studies, as Finkel (2005) and Ackerman (2005) nicely show in their reviews of Sunstein's Risk and Reason (Sunstein 2002) and Lomborg's Global Crises, Global Solutions (Lomborg 2004). A lively and often partisan discussion has taken place on the costs and benefits of measures, with as an extreme a much criticized survey study by Tengs and colleagues (1995), which has spurred volumes of discussion. In that study, actually implemented life saving measures ranged from negative and zero cost per life year saved up to \$20 billion, with cost in many domains lying above one million dollars per life year saved, and an overall median of around \$2000. By focusing on the actual current cost of emission reduction instead of the evaluation of environment and health impacts, some of these uncertainties may be reduced (Oka et al. in this book).

Stated individual preferences

The third approach, based on willingness to pay, is most widely used by economists and most widely despised by no economists. Its strength is that it fits in well to the general approach to economics based welfare analysis, in the dominant Pareto tradition. Its strength is in areas where a comparison with private decisions can easily be made, as in risk of acute toxicity, which individuals may compare to their own occupational risks and their risks taken, for example, in car transport and sports activities. When a clear link to morbidity and mortality is lacking, the link to environmental interventions may not as easily be established by an individual in monetary terms, or even in terms of preference ordering. This is the case, for example, with climate instability, where small-chance high-impact health effects are involved, on possibly long time scales. For broad surveys on this subject, see publications by Portney and Weyant (1999) and especially Kopp and Portney (1999). Also, in cases where non-health related risks are present, as with end-effects on ecosystems and biodiversity, the willingness-to-pay method breaks down in practice. The literature on limitations of the willingness-to-pay approach is vast. On the supportive side of the willingness-to-pay approach, a good survey of operational results of primary and secondary studies is provided by DEFRA (2004), including hedonic pricing and mixed methods.

Revealed individual preferences

Hedonic pricing, the fourth approach, looks at actual choices, its strength as compared to willingness-to-pay statements, which may reflect socially acceptable answers. The main problem is its limitations in application. In comparing different situations all other relevant variables should be kept constant. This hardly ever is the case with other environmental quality aspects, nor with variables other than environmental ones. For example, jobs or housing locations will always differ in many environmental respects, and also in non environmental ones. For environmental aspects not directly related to private quality of life, the hedonic pricing method cannot be applied. This includes future problems, for example, as related to climate change.

1.9 Combined eco-efficiency score

With the economic score and the environmental score ready for specifying the eco-efficiency of case options, there is one final choice to be made, on scaling. In CBA, the data have a meaning in money value. In all approaches not in monetary terms, or not recognizing monetary results as "real money," any linear rescaling of results may take place that does not in any way alter their meaning. Most case applications have such a scaling step, named normalization, both in decision theory and in LCA impact assessment. Two schools in this area respectively go for case-specific internal normalization and weighting, as is usual in decision theory, and for a "supra-case-level" external normalization, as propagated in LCA (Heijungs et al. 1992; Norris 2001). Because weighting is relative to normalization, the weighting factors are to be adjusted each time the normalization reference is adopted. The internal normalization is relative to the current situation, as done by Suh and colleagues (2005), or to the average score of the options compared, as done by Rüdenauer and colleagues (2005). The external normalization is relative to country level with the United States

(BEES 2004) and Dutch example (Huppes et al. 1997) mentioned previously, or to a global reference (Oers et al 2001 and Huijbregts et al. 2001). It seems that internal normalization and case specific weighting are not easily aligned, leading to a certain vagueness in case results. Also, some internal normalization methods may lead to dependence on irrelevant alternatives, where adding an alternative not chosen leads to a different preference ordering of relevant alternatives; see work by Arrow (1970) and Sen (1970). This is the case if a historical alternative is taken as the basis for normalization, as in eco-efficiency analysis of win-win situations. One further problem of case-specific normalization is that seemingly similar eco-efficiency scores cannot be compared between cases, and in some cases are not even comparable if new relevant alternatives are added, as when the reference is an average of the alternatives studied. A conclusion here is that if external normalization is available, it has clear advantages in terms of comparability of eco-efficiency scores between cases.

Further issues in implementation

The analytic framework is to be used in practice, filling it in with data on alternatives. In many decision situations, however, the required technology specifications are not available. Most policy instruments give only indirect guidance on development of technologies and products; in a design stage, specifications for eco-efficiency analysis are lacking; and major investment decisions usually involve larger numbers of technologies, which at least partly require further development before detailed eco-efficiency analysis may become available. In such situations, proxy variables may be used, related to aspects determining the ultimate eco-efficiency, and procedures can be developed for guiding actions toward eco-efficiency, as specified, for example, by Möller and Schaltegger (2005). This often will involve the knowledge and wisdom of experts. It also becomes of paramount importance to monitor past developments of eco-efficiency and its constituent parts, as therein lies the growth and validation of such expert wisdom. This monitoring first involves the performance of larger units such as firms, sectors, and regions. As in eco-design, though, replaying decisions with detailed hindsight would also constitute extremely useful exercises.

1.10 Summary and conclusions

Why eco-efficiency?
To meet the challenge of combining increased affluence with corresponding environmental quality, micro-level choices on the environment economy trade-off have to be aligned to macro level requirements. Practical measures of eco-efficiency are required, and mainly lacking.

What subjects for eco-efficiency?
Three basic situations may be discerned where eco-efficiency for decision making can be applied, each with totally different outcomes. Marginal eco-efficiency, as trade-off between potentially optimal alternatives, is most basic for decision making and can be applied at both the micro and macro level, whereas incremental eco-efficiency at micro level can be translated into environmental effectiveness at macro level. The third, use of the win-win type of eco-efficiency, seems not useful and even confusing.

Economic score
For the economic part of the eco-efficiency ratio, there are three basic approaches available, all based on life-cycle costing: market cost related values, as in management accounting and budget cost analysis; cost-benefit analysis, for the market related cost and benefits; and a steady state type of cost, conceptually best linked to steady state models for environmental analysis such as LCA. Establishing the economic score raises no fundamental problems, but several practical ones, for example, as related to discount rates and to mechanisms to take into account in the analysis.

Environmental score
For the environmental score, there is lessconsensus on what constitute relevant environmental impacts and what are adequate models for their empirical analysis, and on how different types of environmental effects may be combined into a single score. For the modeling of effects, a divergence arises with regard to relatively well established midpoint empirical modeling, linking emissions and other environmental interventions to environmental effects such as climate change and eutrophication, and more speculative endpoint models, linking environmental interventions to health effects, effects on ecosystems, and effects on production functions.

Discounting problems, as present in economic analysis, are even more prominent in environmental analysis due to the long time horizons of many environmental effects. Discounting also is difficult to reconcile with major sustainability considerations on intergenerational justice. Even if modeling choices are accepted, there are four or five fundamentally different options for combining effects as modeled into a single score. Keeping modeling and aggregation scores clear and explicit seems a minimum requirement, often not yet met.

Combined score

It is very common to transform the economic or the environmental score into a case-specific normalized score, in line with customary approaches in multi-criteria or multi-attribute decision theory. This practice deletes the information necessary for optimality analysis, as is required in comparing attractiveness of investments in different technology domains and in linking micro-level decisions to macro-level effects.

Applications

Similarly to the discussions on ecodesign and LCA, we may distinguish between actual decision support, often difficult because of lack of data, and the proxies and procedures used to guide decisions toward the desired eco-efficiency. Quantified eco-efficiency scores will be possible only at a certain nearly final stage of design. Historical studies, both on decision situations and on performance of larger units such as firms, sectors, and countries, would be very useful to build up expert knowledge on eco-efficiency of as yet vaguely defined situations, as is the case with most environmental policies and early stages of larger investment plans. One easy step toward better comparability of studies in different domains of application is not to rescale the environmental and economic scores relative to a case-specific option.

Prospects

A final environmental effect model will never exist, nor a fully agreed on method of aggregation of different environmental effects. Nor will full agreement be reached on details of establishing the economic score. Even so, disagreement is not so fundamental that scores could not be established with a reasonable level of acceptance, especially if it is shown how results depend on assumptions. Such transparency is lacking due to a lack of an explicit framework for eco-efficiency analysis. Agreement on such a framework, as proposed here, and consensus formation on main approaches for quantification is a clear task ahead, essential for realizing a better environment. One essential area of application is in the design of new technologies and products, as it is in this domain of *eco-innovation* that main environmental improvements will have to be realized.

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General Methods

2 Maximum abatement costs for calculating cost-effectiveness of green activities with multiple environmental effects

Tosihiro Oka[a], Yoshifumi Fujii[b], Masanobu Ishikawa[c], Yu Matsuno[d] and Shu Susami[e]
[a]*Graduate School of Economics and Administration,Fukui Prefectural University, 4-1-1 Kenjojima Matsuoka-cho, Japan*
[b]*Bunkyo University*
[c]*Kobe University*
[d]*Meiji University*
[e]*Ebara Corporation*

2.1 Introduction

We have proposed a Maximum Abatement Cost (MAC) method as a means of assessing preferential purchasing with multiple environmental effects (Oka et al., 2005). The MAC method allows assessment of the cost-effectiveness of introducing a product with less emissions of some pollutants than conventional products. In the MAC method, the reduction of a pollutant is multiplied by the MAC, the maximum unit cost of the measures taken elsewhere in society to reduce the pollutant, and is added up over the relevant pollutants. The total sum, called Avoidable Abatement Cost (AAC), is compared with the additional private cost of the product for the purchaser. When the additional private cost is smaller than the AAC, the product is regarded as relatively efficient.

Our previous article (Oka et al. 2005) described the MAC method in detail, as well as presenting an application of the method, and discussing differences between the MAC method and several existing weighting methods for life cycle assessment (LCA), along with the advantages and limitations of the MAC method. The purpose of this article is to provide

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a basis for calculation and report some results on the estimated maximum cost of reducing the environmental burden for various parameters required for this method. The Mac values are presented for nitrogen oxides (NO_x), sulfur dioxide (SO_2), carbon dioxide (CO_2), particulate matter (PM), theoretical oxygen demand (TOD), trichloroethylene (TCE) and perchloroethylene (PCE), heavy metals (HM), volatile organic compounds (VOC) and dioxins (DXN).

This paper first presents an explanation of the concept of MAC, as far as necessary for the following description of the estimated MAC values for the individual parameters. Next, the estimations are described in detail.

2.2 Maximum Abatement Cost (MAC)

MAC is defined as the highest unit cost, i.e., the cost per kilogram of emission reduction, of the activity that has the highest unit cost among all the activities carried out or expected to be carried out shortly to reduce the emission of a substance or a group of substances, here specified for Japan. These emission reducing activities are carried out on the basis of decisions by people and industries with the aim of complying with government regulations, earning a good reputation or obeying their own moral belief.

In order to determine the MAC, the activity incurring the highest unit cost must be specified. Strictly speaking, when a particular person or company is carrying out an activity with exceptionally high unit reduction cost, this very high unit cost may be adopted as the MAC value. Also, when an activity can be divided into several parts that have different unit costs, the partial activity with the highest unit cost may be adopted as the one representing the MAC. However, it is difficult to identify a small activity with very high unit cost contributing only a tiny part of the emission reduction, and it would not be appropriate to use the value for an exceptional activity as a reference value for assessing other activities. For these reasons, we have determined the MAC values from data on the unit cost of emission reduction for activities widely adopted in society.

One problem with the MAC method is how to allocate the cost of an activity that reduces emissions of several substances. We avoided this problem by identifying activities that predominantly reduce only one substance or one impact category, but this problem has remained for TOD, TCE and PCE. All the MAC values, except that for CO_2, are based on cost data for the reduction activities that have actually been carried out. For CO_2, we used estimates of abatement cost to meet the Kyoto target, as specified by the Central Environmental Council (Tyuuou-Kankyou-Singikai Tikyuu-Kankyou-Bukai 2001).

Measuring MAC involves assessing the ratio of cost and emission reduction. If the time of incurrence of cost does not coincide with that of the occurrence of reduction, it is justifiable to account for this difference by time discounting. Thus, if the discount rate changes, the results will change. In the present paper, based on the estimation of real interest rate for recent years (Oka 1999), a 3% discount rate is applied, unless otherwise stated.

2.3 NO_x

2.3.1 MAC related to legally mandated automobile NO_x controls

Legally mandated automobile NO_x controls

Since meeting the environmental standard on nitrogen dioxide (NO_2) in metropolitan areas was poor under existing Japanese regulations, new measures to reduce nitrogen oxides in motor vehicle exhaust gas emissions were introduced in 1992 as the `Special Measures Law on Reduction of Total Emissions of Nitrogen Oxides from Automobiles in Designated Areas (Automobile $NO_x$ Law)'. This law was amended in 2001 and was renamed `Special Measures Law on Reduction of Total Emissions of Nitrogen Oxides and Particulate Matter from Automobiles in Specified Areas (Automobile NO_x, PM Law)'. In order to obtain the MAC for NO_x reduction only, we here estimate the NO_x reduction under the old NO_x Law and the cost of this reduction.

One of the key elements of this law was that for areas where it had been difficult to meet the environmental standard for nitrogen dioxide with conventional control measures (the so-called designated areas), emissions limits for vehicles used in these areas (designated vehicles) were set at stricter levels than previous emission limits. The designated areas were the Tokyo area and the Osaka area[1]. Designated vehicles included trucks, buses and specially permitted commercial vehicles based in the designated areas. An emissions standard equivalent to the strictest standard for each gross vehicle mass class under the Air Pollution Control Law was applied to these designated vehicles. A key point is that the same standards were applied to both gasoline and diesel vehicles. Another feature of the regulations under this law was that they were applied not only to newly produced or registered vehicles but also to vehicles that were already being used. Designated vehicles that did not meet these standards would not pass the mandatory vehicle inspections and could not be used. However, a grace period was included in view of the large impact of these regulations. In addition to the grace period, a notice period and severe change mitigation period have been defined. Thus, the actual application to operating vehicles for regular trucks is shown in Table 2.1.

Even with the grace period, notice period and severe change mitigation period, once these periods are exceeded, not being able to continue the use of vehicles that do not meet the standard means a cost burden to the vehicle user. This should be measurable in terms of the loss to the user caused by the compulsory shortening of the amortization period.

Table 2.1 Application of regulations for in-use vehicles

Year	Vehicles that cannot be used
1995	Vehicles registered in 1984 or before, not meeting the emission standards for designated vehicles
1996	Vehicles registered in 1985 to 1987, not meeting the emissions standards for designated vehicles
1997	Vehicles registered in 1988, not meeting the emissions standards for designated vehicles
1998~	Vehicles registered since 1989, not meeting the emissions standards for designated vehicles

[1] Later, the Aichi and Mie regions were added to the designated area by the Automobile NO_x and PM Law.

Cost related to tightening NO_x regulations

If the price of the vehicles is P, the number of years of shortening is t and the time discount rate is *i*, the cost is expressed as:

$$P(1 - e^{-it}) \tag{2.1}$$

The number of years of shortening is given by the average number of remaining service years in the absence of the regulation. This is estimated as follows.

The second column of Table 2.2 shows the scrapping rate for standard diesel trucks, with vehicle age in 1994. The vehicle age 1.25, for example, has a value of 0.86%. This means that 0.86% of the standard diesel trucks aged 0.25 year (3 months) to 1.25 year (1 year and 3 months) in

Table 2.2 Scrapping rate and average remaining service life of standard diesel trucks by age class (1994)

Vehicle age	Scrapping rate[a]	Remaining service life[b]
0.25	0.86%	11.60
1.25	0.96%	10.69
2.25	1.01%	9.79
3.25	1.54%	8.87
4.25	2.52%	7.99
5.25	4.60%	7.17
6.25	8.14%	6.47
7.25	10.70%	5.96
8.25	13.98%	5.55
9.25	16.79%	5.29
10.25	17.86%	5.16
11.25	17.07%	5.06
12.25	17.28%	4.90
13.25	17.42%	4.71
14.25	18.10%	4.49
15.25	17.44%	4.27
16.25	17.03%	3.96
17.25	-	3.56
18.25	-	3.09
19.25	-	2.52

[a] Calculated from Zidousya-Kensa-Touroku-Kyouryokukai (1994, 1995).
[b] The vehicle scrapping rate after vehicle age 17.25 years is assumed to remain constant.

March 1994 were scrapped during the 1 year period. This value was calculated based on data on the number of registered motor vehicles (Zidousiya-Kensa-Touruku-Kyoukai 1994, 1995). Based on these scrapping rates, the average remaining service life for each vehicle age can be calculated from the following formula:

$$L(t)=\Sigma_{k=t}^{T} s(k)/s(t)$$

where $L(t)$ is the average remaining service life for vehicle age t, T is the maximum, and $s(k)$ is the rate of remaining in service at vehicle age k, which is defined by the following:

$$s(0)=1$$

$$s(k)=s(k-1)[1-d(k-1)]$$

where $d(k)$ is the scrapping rate at vehicle age k. The third column of Table 2.2 shows the average remaining service life by vehicle age based on the 1994 scrapping rate.

As shown in Table 2.1, vehicles that could not be used as of 1995 were vehicles registered up to 1984. This is equivalent to a vehicle age of 10.25 years or over. The average remaining service life of these vehicles can be obtained by weighting the average value for remaining service life for vehicles aged 10.25 years and older from Table 2.2. The number of registered diesel trucks by age class in 1995 (Zidousya-Kensa-Touruku-Kyouryokukai 1995) was used as a weighting factor. Vehicles that could not be used as of 1996 were those aged between 8.25 years and 10.25 years. The average remaining service life was obtained by the same method. Vehicles that could not be used as of 1997 were those aged 8.25 years. Table 2.3 shows the average remaining years of service life for these vehicles from 1995 to 1999.

Table 2.3 Average remaining service life for vehicles scrapped due to the automobile NO_x law

1995	1996	1997	1998	1999
4.49	5.38	5.55	5.55	5.55

Table 2.4 Standard diesel truck average price (Yen/truck)

1995	1996	1997	1998	1999
2,549,756	2,472,581	2,205,592	1,927,197	1,871,119

Table 2.5 Cost of service life reduction for standard diesel trucks (yen/truck)

1995	1996	1997	1998	1999
321,235	368,726	338,392	295,679	287,075

Table 2.6 Scrapping age of small trucks and buses

	1995	1996	1997	1998	1999
Small trucks	> 9.25	7.25-9.25	7.25	7.25	7.25
Buses	> 12.25	10.25-12.25	10.25	10.25	10.25

Table 2.7 Cost of NO_x reduction for small trucks and buses (yen/vehicle)

	1995	1996	1997	1998	1999
Small trucks	127,132	148,424	162,967	173,746	180,448
Buses	435,444	468,880	470,850	463,492	462,008

The average price for a standard diesel truck, taken from the `Machinery Statistics Annual' (Tuusyou-Sangyou-Daizin-Kanbou-Tyousa-Toukeibu, 1995-1999) is shown in Table 2.4. From the above data, the cost of the reduction of service life can be calculated per truck, according to formula (2.1), assuming a 3% discount rate. The results of this calculation are shown in Table 2.5.

The same method can be used to calculate the cost of reducing the service life for small trucks and buses. The age at which small trucks and buses can no longer be used differs from that for standard trucks and is shown by year in Table 2.6. For buses, the median between large buses and microbuses was taken. The cost of service life reduction for small trucks and buses per unit is shown in Table 2.7.

Amount of NO_x reduction

When designated vehicles not meeting the emissions standards are scrapped, it is anticipated that these would be replaced by vehicles meeting the standard. Changes in NO_x emissions were estimated as follows.

Table 2.8 shows the NO_x unit emissions in 1996 for standard trucks, small trucks and buses from the Environment Agency's `Report on Vehicle Unit and Total Emissions' (Kankyoutyou 1998). For medium and light standard trucks and buses and small trucks, the emissions standards for designated vehicles were set to be the same as for recent gasoline vehicles. The scrapping of vehicles not complying with the standards has resulted in an assumed reduction of annual per vehicle NO_x emissions of 112.5-35.0=77.5g for standard trucks, 12.1-7.4=4.7g for small trucks and 103.8-37.1=66.7g for buses.

For heavy standard trucks and buses, the emissions standards for designated vehicles were set at the same level as for recent diesel vehicles. Emissions were estimated based on the changes in emission standards over time, and the reduction rate for scrapped vehicles by initial registration year is shown in Table 2.9. The average NOx reduction rate for each scrapping year was calculated as a weighted average of the values in Table 2.9 taking the number of the heavy vehicles with each initial registration year as a weightig factor, as in the above cost calculation. The resulting average NO_x reduction for each scrapping year is shown in Table 2.10.

The unit NO_x emissions prior to scrapping were assumed to be the same as for the diesel vehicles in Table 2.8, and the reduction rates from Table 2.10 were applied to obtain the reduction rate.

Table 2.8 NO_x emissions (1996, units: kg/vehicle/year)

	Diesel	Gasoline
Standard trucks	112.5	35.0
Small trucks	12.1	7.4
Buses	103.8	37.1

Table 2.9 NO_x emission reduction rate due to replacement of heavy vehicles

Initial registration year	Reduction rate	Initial registration year	Reduction rate
1988	14.3%	1981	35.7%
1987	14.3%	1980	35.7%
1986	14.3%	1979	35.7%
1985	26.5%	1978	47.1%
1984	26.5%	1977	47.1%
1983	26.5%	1976	55.0%
1982	35.7%		

Note: Calculated assuming a 50:50 split between auxiliary chamber and direct injection types, based on Environment Agency Air Quality Bureau (1994), p7, NO_x Reduction Effects of Automobile Exhaust Regulations.

Table 2.10 NO_x emission rate for heavy vehicles by year of scrapping

	1995	1996	1997	1998	1999
Standard trucks	35.5%	26.5%	14.3%	14.3%	14.3%
Buses	38.9%	26.5%	26.5%	26.5%	14.3%

Table 2.11 NO_x emission reduction due to scrapping of vehicles not meeting emission standards (kg/vehicle/year)

Year of scrapping	1995	1996	1997	1998	1999
Standard trucks	53.8	47.5	38.8	38.8	38.8
Small trucks	4.7	4.7	4.7	4.7	4.7
Buses	50.1	42.0	42.0	42.0	34.0

Table 2.12 Present value of NO_x reduction due to scrapping of designated vehicles not meeting emission standards (kg/vehicle)

Year of scrapping	1995	1996	1997	1998	1999
Standard trucks	226.1	236.0	198.4	198.4	198.4
Small trucks	19.3	21.7	22.4	22.4	22.4
Buses	159.6	165.9	177.7	177.7	143.9

According to the Environment Agency's report (Kankyoutyou 1998), the contribution of heavy standard trucks and buses to the total emissions is 63%. The combined average reduction for medium and light vehicles and heavy vehicles for NO_x emissions is shown in Table 2.11. Here, the average remaining service life from Table 2.3 was used to obtain the present value of the total emission reduction per vehicle during the remaining service life period, as shown in Table 2.12. The present value of the total emission reduction was calculated by the following equation:

$$\int_0^T Qe^{-it}dt=Q/i\,(1-e^{-it}) \tag{2.2}$$

Here Q is the annual emission reduction per vehicle, *i* the annual discount rate, *T* the remaining service life.

NO_x unit price for NO_x emission reduction and MAC

The unit reduction cost for all vehicles can be obtained using the ratio of the values in Tables 2.5, 2.7 and 2.12, multiplied by the number of scrapped vehicles. The data and results are shown in Table 2.13. The number of scrapped vehicles in this table was obtained from the year of initial registration, assuming that the ratio of the number of diesel vehicles with each year of initial registration to the total number of registered diesel vehicles in the designated area is the same as that for the whole country. Based on these results, the unit price for NO_x emission reductions is between 2.13 and 2.68 million yen/ton.

Table 2.13 Unit cost of NO_x reduction due to scrapping of designated vehicles not meeting emission standard (10,000 yen/ton)

		1995	1996	1997	1998	1999
Scrapped vehicles	Standard trucks	94,822	78,912	44,724	50,854	55,462
	Small trucks	169,268	181,131	84,336	85,812	85,890
	Buses	6,899	7,583	2,841	2,956	3,283
Cost (100 million yen)	Standard trucks	305	291	151	150	159
	Small trucks	215	269	137	149	155
	Buses	30	36	13	14	15
	Total	550	595	302	313	329
NO_x reduction	Standard trucks	21437	18622	8873	10089	11003
	Small trucks	3,274	3,932	1,892	1,925	1,927
	Buses	1,101	1,258	505	525	472
	Total	25,812	23,812	11,270	12,540	13,402
Unit cost (10,000 yen/t)		213	250	268	250	246

2.3.2 Other NO_x control measures

2000 Control regulations for gasoline vehicles

The cost per ton of NO_2 reduction to meet the 2000 regulations for gasoline vehicles is estimated to be 2 million yen (Nihon-Sougou-Kenkyuusyo 1998).

Stationary source NO_x control

The construction cost of flue gas NO_x control systems for xxx ammonia catalytic reduction (dry method), which have a high NO_x reduction efficiency in electric power plants, is reported to be 3,000 to 6,000 yen/kW (Nihon-Sangyou-Kikai-Kougyoukai 1993). From this data, the cost per ton of NO_2 reduction is estimated to be 120,000 to 170,000 yen. The cost of NO_x control for sintering furnaces and coking ovens at steel works near urban areas and new coal-fired power plants is assumed to be 6,500 to 8,000 yen per kW (Ando, 1990), the cost per ton of NO_2 reduction being 200,000 to 280,000 yen. The estimated NO_x control costs per ton of NO_2 reduction for a sintering furnace at a steel works, based on equipment investment costs of 5.6 billion yen (total investment cost for construction), running costs of about 200 million yen and an annual NO_x reduction of 2,000 tons, is estimated to be 270,000 to 300,000 yen.

2.3.3 MAC for NO_x reduction

Based on the above estimation, the MAC for NO_x reduction is estimated to be 2.5 million yen/ton or 2,500 yen/kg.

2.4 SO_x

2.4.1 SO_x emissions control methods

Sulfur oxides are a traditional air pollutant derived from the sulfur in crude oil, heavy coal and other fuels and raw materials for steel production. When these sulfur-containing materials are combusted, sintered, etc., the sulfur reacts with the oxygen in the air and sulfur oxides are formed. The following equation is useful in understanding the method for reducing SO_x emissions from a given source.

$$SO_x \text{ emission} = \text{production x (fuel consumption/production) x } (SO_x/\text{fuel}) \text{ x } (SO_x \text{ emission}/SO_x \text{ generation}) \qquad (2.3)$$

Japan began to implement measures to reduce SO_x emissions from the second half of the 1960s. By the second half of the 1970s, SO_x emissions had been reduced to the point that almost all areas of the country were meeting the environmental standard. Equation (2.3) suggests that the measures to reduce SO_x emission are classified into four categories: (i) production reduction, (ii) (fuel consumption/production) reduction, (iii) (SO_x /fuel) reduction, and (iv) (SO_x emission/ SO_x generation) reduction.

Production reduction belongs to the first category, energy conservation to the second, use of low sulfur fuel to the third, and flue gas desulfurization and use of fluidized bed boilers to the fourth category.

Of these measures, those that substantially contribute to reducing pollution include energy conservation, fuel switching and flue gas desulfurization. Reduced production was a measure applied in response to local government request to address emergency conditions and/or address short-term local pollutant concentration reduction. There were also a number of cases where a given site had become the subject of attention and the production facility was relocated domestically or internationally. However, from an LCA perspective, this cannot be said to be a reduction of pollution. Additionally, production reductions have occurred during economic recessions on an unplanned basis. The approach of reducing fuel consumption/production means improved efficiency in fuel consumption, and has probably been implemented in many cases. However, no information is available on the quantitative relationship between this approach and SO_x emission reductions. Although energy conservation can greatly contribute to SO_x emission reductions, the net costs of energy conservation would have been less than zero in many cases and it is uncertain to what degree SO_x emission reductions were the objective. In other words, it is unclear how much of a motivating factor the need for SO_x emission reduction was in achieving the progress in energy conservation measures. On the other hand, reducing the sulfur content of fuel and flue gas desulfurization are measures that were implemented with the specific objective of SO_x emission reduction, and have contributed greatly to such emission reductions (Purozyekuto Nyuusu Sya, 2001). Fluidized bed boilers and circulating fluidized bed boilers were introduced from the mid 1980s primarily for coal-fired boilers. As desulfurization occurs in the boiler, the use of flue gas desulfurization is unnecessary. However, the use of this technology has not increased as much as was originally anticipated. On the basis of the above observation, we concentrate on two measures: use of low sulfur fuels and flue gas desulfurization.

2.4.2 Use of low-sulfur fuels

Table 2.14 shows recent fuel prices for the same heating value. If LNG is excluded, it can be seen that fuels with low S content tend to have higher prices. The low price of electric power and LNG is thought to be the result of long-term contracts. One of the causes is also that LNG is imported in massive amounts. This is supplied to other industries in the form of gas at somewhat higher prices than heavy oil and other fuels. Also, power plants that can burn crude oil can achieve SO_x reductions at lower cost than those using heavy oil with reduced sulfur content, although this is not shown, since precise data cannot be obtained[2].

Let us calculate the marginal abatement cost for the case where an existing facility switches fuels to reduce SO_x. This is appropriate for a facility that can only use the fuels listed in Table 2.14. One type of facility meeting these criteria are heavy oil fired thermal power plants. As the fuel oil for thermal power plants is C heavy oil, price differences related to the sulfur is the marginal abatement cost. This is shown in Table 2.15.

It is clear that the unit reduction cost increases gradually with decreasing sulfur content. During this period, the marginal reduction cost for a heavy oil fired thermal power plant using C heavy oil with 0.1% sulfur content was about 312 yen/kg-S[3]. The average unit cost of reducing S from 3% to 0.1% is 156.8 yen/kg-S, almost half of the marginal cost.

2.4.3 Flue gas desulfurization

Flue gas desulfurization was introduced in about 1970, mainly for boilers. More recently, desulfurization installations for waste incinerators have increased and currently account for about 40% of installations, although the treatment capacity is no more than 10% of total capacity. Although boiler installations account for less than 30% of installations, these account for about 70% of treatment capacity (Purozyekuto Nyuusu Sya 2001).

[2] Matsuno and Ueta (1997) shows this using the case of a power plant of the Kansai Electric Power Company

[3] It was 5,376 yen/kg S content in 1980, as stated in the footnote to Table 2.14

Table 2.14 Fuel price for the same heating value and sulfur content

	S Content [Weight %]	Specific Gravity	Price [yen/kl]	Heatinging Value [kcal/l]	S content [kg/10^6 kcal]	Cost [yen/ 1000kcal]
C Heavy Oil HPP	0.2	0.93	17540	9800	0.190	1790
General Sales Jan-Mar 1999	1.5	0.93	15000	9800	1.423	1531
C Heavy Oil LPP-	0.3	0.93	19840	9800	0.285	2024
General Sales	1.5	0.93	15200	9800	1.423	1551
Jan-Mar 1999	1.6	0.93	14800	9800	1.518	1510
	2.6	0.93	13800	9800	2.467	1408
	3.0	0.93	13600	9800	2.847	1388
C Heavy Oil HPP-	0.2	0.93	19110	9800	0.190	1950
General Sales Apr-Jun 1999	1.5	0.93	16220	9800	1.423	1655
C Heavy Oil LPP-	0.3	0.93	21440	9800	0.285	2188
General Sales	1.5	0.93	16420	9800	1.423	1676
Apr-Jun 1999	1.6	0.93	15800	9800	1.518	1612
	2.6	0.93	14800	9800	2.467	1510
	3	0.93	14600	9800	2.847	1490
C Heavy Oil	0.1	0.93	17830	9800	0.095	1819
ElectrUtilityity	0.2	0.93	17540	9800	0.190	1790
General Sales	1.6	0.93	14800	9800	1.518	1510
Jan-Mar 1999	2.6	0.93	13800	9800	2.467	1408
	3	0.93	13600	9800	2.847	1388
Kerosene Large Steel Company Sales Jan-Mar 1999	0.02	0.79	27000	8900	0.018	3034
Kerosene Large Steel Company Sales Apr-Jun 1999	0.02	0.79	29000	8900	0.018	3258
L/S A Heavy Oil Large Steel Comp. Sales Jan-Mar 1999	0.1	0.83	27000	9300	0.089	2903
L/S A Heavy Oil Large Steel Comp Sales Apr-Jun 1999	0.1	0.83	29000	9300	0.089	3118

Table 2.14 (cont.)

Table 2.14 (cont.)

A Heavy Oil Large Steel Comp Sales Jan-Mar 1999	0.5	0.85	26000	9300	0.457	2796
A Heavy Oil Large Steel Comp. Sales Apr-Jun 1999	0.5	0.85	28000	9300	0.457	3011
Propane Large Steel Comp. Sales Jan-Mar 1999	0.01		41400	12000	0.008	3450
Propane Large Steel Comp. Sales Apr-Jun 1999	0.01		37800	12000	0.008	3150
LPG Osaka Gas Mar 1999	0.01		30290	12000	0.008	2524
LNG Osaka Gas Mar 1999	0.001		18000	13000	0.001	1385
LNG Tokyo Electric Power FY1998	0.001		20614	13000	0.001	1586
LNG Kansai Elcetric Power FY1998	0.001		19797	13000	0.001	1523

Source: Sekitu (1999).
Note: The cost differential related to sulfur content has shown a decreasing trend. For example, the cost for the cleanest grade of C heavy (electric power use) 0.1% was about 5,000 yen/0.1% in 1980 and from the table above was 290 yen/0.1%, about 17 times.

Table 2.15 Price difference by S content for electric utility use of C heavy oil (Jan-Mar 1999) S content S content Price difference/S content

S content [Weight %]	**S content** [kg/10^6 kcal]	**Price difference/S content** [yen/kg S content]
0.1-0.2	0.09-0.19	311.8
0.2-1.6	0.19-1.52	210.4
1.6-2.6	1.52-2.47	107.5
2.6-3.0	2.47-2.85	53.8

The Flue Gas Treatment Technology Manual (For Government Agencies) Heisei 3 Environment Agency Contract by the Japan Industrial Machinery Society (Nihon-Kikai-Kougyou-Rengoukai 1992) contains several examples of flue gas desulfurization costs. The average costs of the limestone gypsum method, which is the most common method, were estimated. The input data are shown in Table 2.16.

For a C heavy oil fired boiler, it can be assumed that 10,000 Nm^3 would be handled for 0.85kl. The sulfur content is calculated to be 3.6%, thus assuming a fairly dirty fuel in the calculation. With due caution (as the data relates to different periods), it can be calculated that desulfurization to 0.4 kg/million kcal can be done for 86 yen/kg-S. A comparison with Table 2.15 shows that, for this degree of desulfurization, the cost of using flue gas desulfurization equipment is less than that of using low-sulfur fuel oil.

This is the reason for the large number of flue gas desulfurization units that have been installed (2,075 as of March 1998).

The Japan Research Institute (Nihon-Sougou-Kenkyuusyo 1998) has calculated the average cost of flue gas desulfurization by compiling data on cumulative investments, etc. (see Table 2.17) This is the cost data for emission sources and it is possible to compare this with our data. Their data shows that 2.6667 million tons of SO_2 (1.3335 million tons as sulfur) were removed by flue gas desulfurization. The average cost was 104.9 (yen/kg S), which is close to our results. In addition, their study used the shortened legal service life (7 years). If a period of 14 years had been used, an excess of 19.4 billion of amortization would have been included. This adjustment yields a value of 90.4 (yen/kg S), which is almost the same as our results. Their research yielded the overall average cost of flue gas desulfurization, and the results do not necessarily have to agree with our work, which is based on point data. However, it is likely that the average cost of SO_x reduction by flue gas desulfurization is roughly at this level.

In a mix with fuel sulfur reduction, the sulfur removal by the flue gas desulfurization is equal to the fuel sulfur reduction x (1-desulfurization efficiency). Thus, the SO_x emissions are reduced by less than the amount of fuel sulfur reduction. The gradual increase in cost is magnified and the efficiency of sulfur removal is reduced by the reduction of the fuel sulfur content. In other words, it can be anticipated that this method would

Table 2.16 Cost and other data for flue gas desulfurization

	Facility cost 1000yen/(Nm^3/h)	**Running cost 1000yen/year/ (Nm^3/h)**	**Efficiency %**
Limestone-gypsum method	4-8	0.83	90-95
Conditions for calculating running costs:			
Capacity	660,000 Nm^3/h (200MW)		
SO_2 concentration	2,000 ppm		
Desulfurization rate	90%		
Operation time	6100 h/year		

Table 2.17 Estimation of average cost of flue gas desulfurization

	Cost	
Annual oparating cost	547,800	1000yen/year
Facility cost[a]	3,960,000	1000yen
Facility cost annualization[b]	340,354	1000yen/year
Total (A)	888,154	1000yen/year
	Sufur reduction	
Annual flue gas discharge	4,026,000,000	Nm3/year
Amount SO_2	8,052,000	Nm3/year
Amount SO_2 removed	7,246,800	Nm3/year
Converted to kg sulfur (B)	10,352,571	kg/year
	Average cost (A/B)	
	85.79	yen/kg

[a] Equipment cost is 6000 yen/(Nm3/h).
[b] Facility cost is annualized according to $S=Cr(1+r)^{n-1}/[(1+r)^n-1]$, where S is the amount value, C is the equipment cost, r is the service life, and n is the depreciation period with r=0.03 and n=14 [years].

result in increased maximum cost of reducing SO_x emissions as SO_x emissions decrease.

2.4.4 MAC for SO_x Reduction

The estimated maximum costs are 85.79 yen/kg S for flue gas desulfurization and 312 yen/kg S for heavy oil desulfurization, and the average cost is 156.8 yen/kg S. Taking the lower flue gas desulfurization cost of 85.79 yen and converting kg S to kg SO_2 yields a MAC for SO_x reduction of 43 yen/kg.

2.5 CO_2

With regard to the MAC for CO_2, we have assumed that the emission target for 2008–2012, given by the Kyoto Protocol to the United Nations Framework Convention on Climate Change, i.e. a 6% reduction relative to 1990, will be met, taking into account the COP7 agreement about the inclusion of forest sinks and the use of Kyoto Protocol Mechanisms.

There are two methods to estimate the MAC for CO_2. One approach involves estimating the direct cost of each policy or technology. The other approach is an indirect estimate based on estimated CO_2 emission demand coefficients. The former approach involves the cumulative estimating costs of CO_2 reduction by individual technologies. The method with the highest reduction cost is taken to provide the MAC. The second approach involves the construction of a model to explain CO_2 emissions by explanatory variables including a logarithm of energy price, the estimated coefficient of which represents the price elasticity of CO_2 emissions. When the ratio of the emissions for the standard year and the target year are determined, and the energy price for the standard year is defined, the elasticity value obtained from the model can be used to obtain the energy price that guarantees that the emission target is met, and the MAC can be derived.

2.5.1 Cumulative Method

The Central Environmental Council [Interim Report of Target Achieving Scenario Subcommittee] (Tyuuou-Kankyou-Singikai Tikyuu-Kankyou-Bukai 2001) examined the reduction potential for domestic CO_2 control technologies (in the wider sense of the term technology, i.e. including measures such as the introduction of summer time) and the relationship between cost and reduction for measures that might be implemented to meet the requirements of the Kyoto Protocol. The equation for calculating the additional cost of reduction is shown below:

$$\text{Increased reduction cost} = \\ \text{reduction cost}(C) - \text{energy mitigation cost}(P) - \text{other profit}(E)$$

The scenario indicates that, assuming the construction of 7 nuclear reactors, CO_2 emissions in 2010 will be 355 million tons of carbon or 108% of the 330 million tons in the 1990 reference year. However, the target that must be achieved in 2010 is a 6% emission reduction relative to 1990, or 310 million tons of carbon. Although it is not explicitly stated in the report, if the forest sink that was permitted at COP7 and the 1.8% credit under the Kyoto Protocol Mechanism are subtracted, this leaves 26.7 million CO_2-C that must be addressed in Japan using various technologies. Converting this to CO_2 yields a value of 98 million tons (in the Interim Report, the analysis is written in terms of tons of CO_2, and this convention will be followed here).

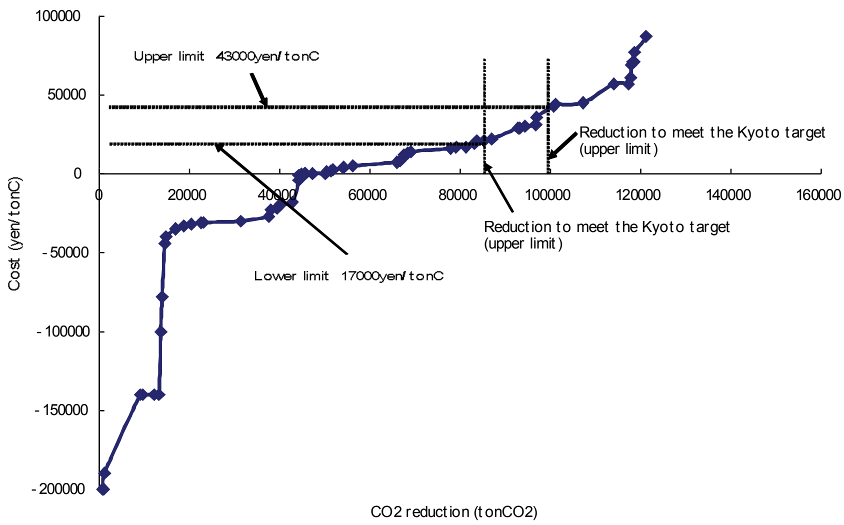

Figure 2.1 Estimated MAC for CO_2 from the Ministry of the Environment's Assessment of Technological Reduction Potential

Figure 2.1 lists the reduction potential and cost of technologies from the Interim Report, in the order from least to greatest cost. Since control technologies include control measures that are either substitutes or supplements, the potential for each of these could not be covered. Therefore, each control measure was considered independently, and the highest reduction cost to meet the target was estimated for two cases:

— the case where inestimable costs are included in the target achievement;

— the case where inestimable costs are all excluded.

The highest cost to meet the reduction target of 98 million tons of CO_2 was 17,000 yen/ton CO_2-C (4,600 yen/ton CO_2) for case 1 and 43,000 yen/ton CO_2-C (11,700 yen/ ton CO_2) for case 2.

2.5.2 MAC from Demand Model

There are a number of examples of estimates of price elasticity values obtained from macroeconomic models. The Ministry of the Environment's report used a bottom-up analysis as well as five macroeconomic and other models to analyze carbon tax simulations. Although these models did not rigorously address the CO_2 emission scenarios discussed above, the models were used to calculate the carbon tax required to reduce carbon dioxide emissions in 2010 to 2% below the 1990 level. As shown in Table 2.18, the carbon tax required to achieve this reduction in the simulation results ranged from 13,000 to 35,000 yen per ton of carbon. Converted to CO_2, this is about 3,500 to 9,500 yen per ton. It should be noted that the model assumed that technology is selected in an economically rational way and that changes in energy price elastically improve energy efficiency. Existing socioeconomic system barriers against individual countermeasures were not addressed.

These results do not differ greatly from the estimation results obtained using the bottom-up method. The results are summarized in Table 2.19.

2.5.3 MAC for CO_2 Reduction

The estimated results yield a MAC for CO_2 reduction of about 7,000 yen/t or 7.0 yen/kg.

Table 2.18 Carbon tax amounts required for the model to achieve a 2% reduction of CO_2 emissions

Model	Case	Amount of carbon tax in 2010 [yen/tonC]
AIM End Use Model	Carbon tax case	30,000
	Carbon tax + subsidy case	3,000
GDMEEN	Carbon tax case	34,560
MARIA	Carbon tax case 1	13,148
	Carbon tax case 2	14,359
SGM	Increase of government expenditure case	20,424
	Government deficit reduction case	21,100
	Income tax refund case	21,080
AIM Material Model	Carbon tax case	15,587

Table 2.19 MAC for CO_2 derived by two methods based on the Central Environmental Council's 2001 estimate

	CO_2 MAC [yen/ton-CO_2]
Bottom-up method	4,600-11,700 (8150)
Demand model	3,500-9,500 (6500)

Note: the values in parentheses are averages of the upper and lower limits.

2.6 SPM (suspended particulate matter)

2.6.1 Introduction

The minute particulate matter (PM) in the atmosphere is generically referred to as floating dust and is measured using methods including light scattering. In Japan, floating dust with a particle diameter less than 10 μm is defined as SPM (suspended particulate matter).

The Japanese government amended the Automobile NO_x-PM Law (Special Measures Law on Reduction of Total Emissions of Nitrogen Oxides and Particulate Matter from Automobiles in Specified Areas) in 2001. Based on this amendment, 276 cities, towns and villages were designated as NO_x and PM control regions, for which PM reduction plans were established by prefectural governors.

The following control measures are expected to be taken to reduce SPM.

1. *Stationary source control measures*

(a) Flue gas emissions sources

i. Waste incinerator measures. To control primary particle emissions from waste management incinerators, the emissions standard for particulate emissions from incinerators with an incineration capacity greater than 200 kg/h was set at 0.04 g/m^{3}N. Also, to control secondary particulate matter formation, the hydrogen chloride emissions standard was reduced to half of the national standard, to 250 mg/m^{3}N for incinerators with an incineration capacity of 200 to 500 kg/h and 100 mg/m^{3}N for incinerators with an incineration capacity greater than 500 kg/h.

ii. NO_x Control Prefectural Ordinance. The Guidance Policy for NO_x Emissions from Factories/Business was enacted as a prefectural ordinance to strengthen the management of combustion sources, etc. and reduce NO_x emissions by 5%.

iii. Strengthening Regulations for Liquid Fuel Boiler Particulate Emission (amendment to prefectural ordinance). Special emissions standards were applied to liquid fuel boilers. Particulate emission standards for liquid fuel boilers with flue gas discharge greater than 200,000 m^{3}N/h were set at 0.04 g/m^{3}N and 0.05 g/m^{3}N for liquid fuel boilers with 40,000 to 200,000 m^{3}N/h flue gas discharge.

(b) VOC control measures

For existing sources, the enacting of a prefectural ordinance based on the Guidance Policy for VOC Control was anticipated to reduce VOC emissions by 20% between 1996 and 2005. For the period from 1990 to 2000, the concentrations of non-methane VOCs decreased by about 20%, and this effect was included.

(c) Small incinerator control measures

With the enforcement of the ordinance, all small incinerators with a capacity of less than 30 kg/h capacity used in homes, etc. were assumed to be removed.

(d) Rice straw burning

Enforcement of the Waste Management Law amendments and prefectural ordinances resulted in a total ban on open air burning. Rice straw burning was originally exempted but was banned as part of the strengthening of control measures. Thus, emissions from rice straw burning are not included.

2. *Mobile source control – enforcement of vehicle regulations from the Amended NO_x Law.*

Enforcement by vehicle type started in April 2002 for new vehicles and from April 2003 for vehicles in use. The long-term standards are assumed to be applied (the vehicles that meet only the short-term standards (the 1988 standards) are due for replacement). Standards for vehicles with capacities of less than 3.5 ton were set at the same level as those for gasoline vehicles. The grace period for small trucks and passenger vehicles was 8 years, while that for standard trucks was 9 years, that for special vehicles 10 years and that for buses 12 years. In this case, the new longer-term regulations were brought forward by two years to 2005 and the diesel fuel sulfur content was reduced from 500 ppm to 50 ppm.

3. *Vehicle control – enacting Ordinance to Strengthen Vehicle Type Control.*

In addition to the vehicle control measures discussed under (2), enforcement of the ordinance for vehicle types led to a shortening of the grace period (8 years for small trucks and passenger vehicles, 9 years for standard trucks, 10 years for special vehicles and 12 years for buses) to a uniform 7 years, and the replacement of vehicles with models meeting new standards was accelerated.

2.6.2 MAC for SPM

The vehicle control system based on the Automobile NO_x - PM Law was estimated to result in a total of 7,215 vehicles being removed from service. When the ordinance to shorten the grace period was introduced, it was estimated that a total of 21,102 vehicles would be removed from service. Available choices include replacement of these vehicles with those meeting regulatory requirements and installation of control equipment.

The cost of control equipment for two companies was investigated, and the results are shown in Table 2.20.

The average cost per vehicle was estimated from the Machinery Statistical Annual Report published by the Ministry of International Trade and Industry (currently Ministry of Economy, Trade and Industry) Ministry Secretariat Statistical Study Department, as shown in Table 2.21 below.

These results show that small trucks and special vehicles were all replaced by new vehicles meeting the latest regulatory requirements. In this case, the price of a new vehicle was taken to be the cost. For standard trucks, installation of control equipment on all vehicles was assumed and the cost per vehicle was assumed to be a uniform 1 million yen. The amortization period for this cost is seven years, with straight-line amortization.

Table 2.20 DPF equipment type and cost

DPF type	Fuel used	Cost
A type of DPF where the material collected by filter is incinerated using external electric power without removing the filter from the vehicle when the vehicle is not in use	Ordinary diesel fuel	800,000 yen
A type of DPF where the material collected by filter is incinerated using external electric power on the filter removed from the vehicle	Ordinary diesel fuel	1,000,000 yen (DPF); 600,000 yen (Regenerating equipment)
A type of DPF using oxidizing catalyst and heat of exhaust gas to combust the materials collected by filter	Low sulfur diesel fuel	1,000,000 yen

Table 2.21 Estimated average cost per vehicle

	Cost [yen/vehicle]
Standard trucks	2,445,114
Small trucks	1,038,705
Buses	4,282,386
Special use vehicles	816,487

Table 2.22 Number of vehicles requiring control measures

		Vehicle control I[a]		Vehicle control II[a]	
		No. of Vehicles	**Cost [10^6 yen]**	**No. of Vehicles**	**Cost [10^6 yen]**
Replacement	Small trucks	194	2,015	438	4,550
	Special use vehicles	78	64	356	291
Control equipment	Sandard trucks	6,924	6,924	20,182	20,182
	Buses	19	19	126	126
Total		7,215	9,022	21,102	25,149
Annualized cost (10^6 yen/year)			1,289		3,593

[a] Under vehicle control I, vehicle type control under the amended Automobile NO_x, PM Law is assumed to implemented, while under vehicle control II, in addition to the vehicle control I, the grace period is assumed to be shortened to 7 years.

Table 2.23 MAC calculation results for SPM concentration reduction

	SPM conc	**Cumulative conc. reduction**	**Cumulative added cost**	**Individual conc. reduction**	**Individual added cost**	**Maximum reduction cost**
	[μg/m³]	**[μg/m³]**	**[10^6yen/year]**	**[μg/m³]**	**[10^6yen/Year]**	**[10^6yen/(μg/m³)]**
1996 base case	52.9	-	-	-	-	-
2001 BAU	42.3	-	-	-	-	-
Control case 1	41.3	1.0	1,289	1.0	1,289	1,289
Control case 2	41.3	1.5	3,593	1.53	3,593	2,395
Control case 3	38.9	3.4	4,502+	1.9	909+	478+

Calculation of MAC for SPM Reduction

Table 2.23 shows the reduction of SPM concentrations and the cost for the control cases. The results show that the highest reduction cost is that for mobile sources: 2.395 billion yen per 1 μg/m³ for shortening the grace period to meet vehicle control regulations. The lowest reduction cost is that for stationary source emissions control: 478 million yen per 1 μg/m³.

The particulate matter emissions from vehicles would be reduced from 2,962 tons to 2,529 tons, a difference of 430 tons of particulate matter. Assuming installation of DPF equipment (1 million yen per unit), case 3 would mean that all standard trucks and buses (20,182 trucks and 126 buses for a total of 20,308 vehicles, see Table 2.22) would have this equipment installed. Assuming a service life for the equipment of 7 years, the simple annual costs for one unit would be 1 million yen/7, assuming that no maintenance cost would be incurred. From the reductions of 430 tons of particulate matter, the average cost of DPF equipment installation would be

$$(1000000/7) \times (20182 + 126)/(2962 - 2519) =$$
$$6.7 \text{ million yen/ton particulate matter.}$$

This is taken as the MAC for SPM reduction.

2.7 Eutrophication-causing substances

2.7.1 Industrial wastewater treatment

Lake Biwa Research Institute has reported data on the costs to reduce the discharge of chemical oxygen demand (COD), total nitrogen (TN) and total phosphorus (TP) to Lake Biwa from industries in 1991 (Sigaken Biwako Kenkyuusyo 1994). According to the report, the emissions of theoretical oxygen demand (TOD), where TOD=3COD+[19.7TN+ 143TP]/2) (Ukita 1982), reduced from 21 factories amounted to 3,096 ton/year, the unit costs of which are shown in Table 2.24 by the cumulative percentage of reduction. Most of the last 7% of the reduction was at a cost of 1,300 to 1,600 yen/kg.

Data from Ebara Corporation on plant waste unit treatment costs is shown in Table 2.25.

Table 2.24 Unit costs of reducing TOD from the industries in the Lake Biwa area

Cumulative percentage of reduction	Unit cost [yen/kg]
29.0%	40
46.7%	43
50.9%	53
71.3%	138
75.1%	176
78.7%	312
79.2%	513
82.5%	538
86.6%	547
87.2%	714
92.4%	772
92.6%	827
93.7%	1,312
96.2%	1,357
99.4%	1,361
99.5%	1,526
99.8%	1,576
99.9%	6,149
99.9%	7,126
100.0%	18,645
100.0%	28,209

Table 2.25 Unit cost of factory wastewater load reduction

Industry	Reduced Load [kg/year]				Annual cost [1000 yen/year]	Reduction cost [yen/kg-TOD]
	COD	TN	TP	TOD		
Livestock processing	78,750	14,700	6,300	831,495	28,365	34
Semiconductor	109,760	49,280	0	814,688	62,757	77
Automobile	0	3,142.6	0	30,945	15,362	496
Total				1,677,128	106,484	63

2.7.2 Reduction of pollutants by sewerage systems in the Lake Biwa area

On the basis of cost estimates for the sewerage system in the Lake Biwa area, also obtained from Lake Biwa Research Institute, we can calculate the unit cost for TOD. The total construction and operating costs of the North East Watershed Sewerage System of Lake Biwa, including projection to 2023, was estimated to be 418 billion yen. Subtracting the savings on the existing costs of human waste treatment and the benefits from being able to use flush toilets, i.e. 115 billion yen and 106 billion yen, respectively, the net cost of improving environmental water quality is 197 billion yen, the present value of which is 159 billion yen under a 3% discount rate.

Table 2.26 shows the allocation of the costs to COD, total nitrogen (TN) and total phosphorus (TP). The costs of construction and maintenance of sewers and 76% of the sewage treatment costs are attributed to COD, while 24% of the sewage treatment costs are attributed to TN and TP. The cost savings from ending existing human waste treatment and the benefits from the ability to use flush toilets are all attributed to COD. Reductions of COD, TN and TP are estimated as net values, subtracting the existing reductions by human waste treatment. The present values of the reductions are shown in Table 2.26. The unit cost for TOD is also shown in the table as 1,700 yen/kg, along with the unit costs for COD, TN, TP and TOD (TN,TP), where TOD (TN,TP) is defined as (19.7 TN+143 TP)/2.

Table 2.26 Unit cost for sewerage system (Lake Biwa North East, projection and actual)

Cost [billion yen]		Reduction		Unit cost	
Total	159	92,348	ton-TOD	1,700	yen/kg-TOD
COD	137	17,059	ton-COD	8,000	yen/kg-COD
TN, TP	23	1,918	ton-TN	5,900	yen/kg-TN[a]
		312	ton-TP	36,000	yen/kg-TP[a]
		41,170	ton-TOD[b]	550	yen/kg-TOD

Note: Data are recalculated from Oka (1992).
[a] The cost allocated to TN and TP was evenly reallocated to TN and to TP to calculate unit costs for TN and TP.
[b] TOD(TN,TP)=(19.7 TN+143 TP)/2.

2.7.3 MAC for TOD

Reducing the wastewater load from factories around Lake Biwa by 99% costs less than 1,600 yen/kg-BTOD, and the unit cost of reducing the wastewater load from the Ebara Corporation factory was less than 1,600 yen/kg-TOD for all units. The unit cost of reducing the domestic wastewater burden was 1,700 yen/kg-TOD. We conclude that the MAC for TOD is 1,700 yen/kg-TOD.

2.8 TCE and PCE

Based on the February 1996 report entitled 'Towards Restoring the Famous Hadano Basin Springs' by the Hadano City Environment Department, we calculated the cost of remediation for trichloroethylene (TCE) and perchloroethylene (PCE). As the methods for treating either substance are basically the same, the average unit treatment costs (treatment costs per amount recovered) for these two substances was also assumed to be the same. The amount recovered and the costs (survey cost, cleanup cost) were compiled, and the average treatment cost (treatment cost per

Table 2.27 Treatment cost for TCE, PCE

Treatment method	No.of cases	Average cost [1000yen/year]		Maximum cost [1000yen/kg]	
		Cleanup cost	Total cost	Cleanup cost	Total cost
In Situ Vacuum Extraction Method	20	1,362	3,138	21,667	31,667
In Situ Gas Aspiration Method	13	177	1,150	1,000	10,000
Excavated Low Temp. Heat Treatment	2	15,053	17,097	30,000	34,000
Excavation Industrial Waste Disposal	4	2,767	5,580	10,000	20,000
Excavation Sealing Treatment	2	6,335	6,335	10,345	10,345
All Treatment Cases	41	2,034	3,583	30,000	34,000

Note: Total cost includes survey and cleanup costs, all excavation treatments are the total of excavation low emperature heat treatment, excavation industrial waste disposal and excavation sealing treatment.

unit recovered) was calculated. For cases where treatment was implemented repeatedly by the same method at the same location, the total was counted as one treatment. In cases where both substances were treated simultaneously, the calculation was made for the total cost of the combined treatment (only the combined treatment is shown, cases where individual treatment amounts could be assessed were treated the same). There were cases where the survey cost and cleanup costs were not distinguished, or where there was no clear and detailed definition of the costs of survey and treatment. The results are shown in Table 2.27. The MAC for treating trichloroethylene and perchloroethylene were found to be 15,053,000 yen/kg.

2.9 Heavy metals

2.9.1 Range of study

The technologies covered in this study are technologies for treating heavy metals in wastewater from plating plants and wastewater from municipal solid waste treatment facilities. The facilities covered, types and concentrations of heavy metals are shown in Table 2.28.

Heavy metal species and concentrations

The heavy metals covered in this study were Fe, Cu, Zn, Ni, Sn, Pb, As, Cd, Cr, Hg and Mo. Concentrations in wastewater ranged from 43 ppm to 1,310 ppm. As shown in Table 2.28, the concentrations in wastewater from electronic parts production facilities vary by a factor of 30. The concentration of heavy metals in the wastewater from the municipal waste treatment plant was 386 ppm and fell within the range of wastewater concentrations from plants manufacturing electronic parts.

Table 2.28 Heavy metal wastewater data

Facility	**Heavy Metals**	**Waste water discharge [m^3/d]**	**Quantity of heavy metals [Kg/d]**	**Waste water concentration [ppm]**
LCD Production	Fe Zn Cu Ni	46.1	22.9	497.0
Electronic Parts Production	Fe Pb Cu Sn	264.0	32.8	124.0
Plastic Plating	Fe Cu Ni	46.1	13.8	300.0
Electronic Parts Production	Ni Sn Pb Cu	752.7	32.0	42.5
Electronic Parts Production	Fe Cu Ni	67.2	88.5	1,310.0
Zinc Plating	Fe Zn	80.0	31.7	396.0
Landfill	Sb As Cd Cr Cu Pb Hg Mo Ni	756.0	291.8	386.0

2.9.2 Estimation method

To assess plating plant wastewater treatment, we interviewed companies producing systems to treat sewage which contains heavy metals, asking about capacity and costs. Operating costs included those of chemicals and other consumable supplies, waste treatment cost, energy cost (electric power), facility amortization and personnel costs. The calculation of cost amortization for facilities and construction assumed 275 operating days per year, 7.2% discount rate, a 7 year amortization period and 10% scrap value for equipment, and a 30 year amortization period and no scrap value for buildings and structures. For municipal solid waste treatment facilities, the calculation of facility construction cost (including water treatment facilities) assumed that the buildings and equipment account for 50% each, with 365 operating days per year. The equation to calculate amortization is shown below.

$$\theta=(1-v_s)(1+r)^n/[\Sigma_{k=1}{}^n(1+r)^k] \tag{2.4}$$

Here, θ, v_s, r and n, are the annual cost rate, scrap value, discount rate and amortization period in years.

2.9.3 Calculation results and comments

Results are compiled in Table 2.29.

The cost of reducing heavy metal levels ranges from 1,766 yen/kg to 20,626 yen/kg, with 20,626 yen/kg being the highest cost. Although the number of samples was relatively small, the data provides several useful pieces of information. In general, costs of end-of-pipe technologies are dominated either by the amount of material discharged or by the amount of material treated. Cases where the pretreatment levels are relatively high compared to the discharge standard fall into the former category, while cases where the pretreatment concentrations are relatively low fall into the latter category. Treatment of waste liquids from plating plants falls into the former category.

A comparison of plants D and E shows that, although the wastewater concentrations differ by a factor of 300, the treatment costs only differ by a factor of 3. The wastewater treatment cost for the municipal waste treatment facility was less than that for plating plants, at 1,766 yen/kg.

Table 2.29 Heavy metal treatment cost

Facility	**Heavy Metals**	**Waste water concentration [ppm]**	**Treatment cost [yen/kg]**	**Labor cost [yen]**
LCD Production	Fe Zn Cu Ni	497	20,626	54.1
Electronic Parts Production	Fe Pb Cu Sn	124	10,793	72.3
Plastic Plating	Fe Cu Ni	300	7,572	61.0
Electronic Parts Production	Ni Sn Pb Cu	43	9,163	76.5
Electronic Parts Production	Fe Cu Ni	1,310	3,192	68.3
Zinc Plating	Fe Zn	396	3,667	55.0
Landfill	Sb As Cd Cr Cu Pb Hg Mo Ni	386	1,766	N.A.

Table 2.30 Regression statistics

Regression results			
	Coefficient	**Standard error**	**t value**
Intercept	6.939	1.140	6.089
Waste water volume	-0.6103	0.2513	-2.428
Concentration	-0.7402	0.2875	-2.575
Regression statistics			
Multiple correlation coefficient R			0.8109
Coefficient determination R^2			0.6576
Compensating coefficient of determination R'^2			0.4864
Standard error			0.2579
Number of samples			7

In order to analyze the scale effects related to the wastewater volume treated and the concentration in the treated discharge, a logarithmic multivariate analysis was conducted, with the cost of treatment for heavy metals (Cost: yen/kg) as an explained variable, and the amount of wastewater per day (Q: m^3/day) and the total heavy metal concentration (C: ppm) as explanatory variables.

Although 7 samples is a small number, the value of t was greater than 2, a result that was anticipated from the sign conditions obtained:

$$\text{Cost} = 10^{6:939} \,.\, Q^{-0:6103} \,.\, C^{-0:7403} \tag{2.5}$$

the regression statistics of which are summarized in Table 2.30.

2.9.4 MAC for heavy metals

Within the range included in this study, the highest cost of reducing heavy metals was 20,626 yen/kg. It was inferred that the MAC was higher than 20,626 yen/kg. A logarithmic regression analysis, using the cost of treatment for heavy metals (Cost: yen/kg) as an explained variable and the amount of wastewater per day (Q: m^3/day) and the total heavy metal concentration (C: ppm) as explanatory variables, showed that the treatment costs rose with wastewater volume by a power of -0.6103 and rose with wastewater concentration by a power of -0.7403 (Equation 2.5). Thus, a noticeable scale effect related to wastewater volume and concentration was observed.

2.10 Volatile organic compounds (VOC)

2.10.1 Range of study

The technologies covered in this study are those for the removal of VOC from the exhaust gas of printing facilities for plastic films or metal cans. The technologies used for VOC treatment are direct incineration, catalyst deodorizer, self-sustained combustion, regenerative deodorizer and solvent recovery, as well as combinations of these technologies. All facilities are used to remove toluene.

2.10.2 Data and estimation method

A major converting company and a major packaging manufacturer provided actual operating data, investments and running costs for a one-year period. Data collected by the Japan Printing Machinery Association (JPMA) was also used.

The costs comprise amortization, utilities and personnel costs. The calculation of amortization cost assumed a 7-year amortization period, 7.2% discounting rate and 10% scrap value. The equation is the same as that for heavy metals. The plant was assumed to operate for 6,000 hours per year.

Because the data supplied by JPMA did not include removal efficiency, we assumed the average removal efficiency of the observed data, which was 63.9%.

Because the running costs of packaging manufacture were more than one order of magnitude higher than other corresponding data, we did not used this data to calculate the MAC. The difference may have been caused by differences in the scope of personnel costs.

2.10.3 MAC for VOC

Calculation results are compiled in Table 2.32. The cost of removing VOC from exhaust gas lies within a range of 24 to 157 yen/kg. The highest cost of 157 yen/kg might be an underestimation, because we assumed the average removal efficiency for data from JPMA. Therefore, the MAC for VOC is higher than or equal to 157 yen/kg.

2.11 Dioxin

According to Kishimoto et al. (2001), 358 billion yen of investments were necessary up to 2002 to meet the emission standards on dioxins. This corresponds to 17.2 billion yen per year, converted to an annual basis.

Adding this to the increase in operation cost of 24 billion yen per year yields an annual cost of 41.2 billion yen. The amount reduced by these measures was estimated to be 2,210 g-TEQ/year. Therefore, the unit cost is 19 million yen/g-TEQ. Based on this, the MAC for dioxin is taken to be 19 billion yen/kg-TEQ.

2.12 Summary

The MAC results obtained are compiled in Table 2.31. These values can be used to assess green activities, an example of which is shown in Oka et al. (2005).

Table 2.31 MAC values (yen/kg)

	NO_x	SO_x	CO_2	SPM	TOD	TCE, PCE	HM	VOC	DXN
MAC	2,500	43	7.0	6,700	1,700	1.5×10^7	20,000	160	1.9×10^{10}

Table 2.32 Cost for reducing volatile organic compounds

Data source	Input [ton/year]	Removal [ton/year]	Investment [1000yen/year]	Running [1000yen/year]	MAC [yen/kg]
converter	170	165	147,291	535	144
converter	239	224	135,000	535	97
converter	650	452	171,500	589	61
converter	962	897	245,431	535	44
converter	388	255	115,000	10,657	113
packaging	241	44	26,531	23,226	-
packaging	510	235	196,260	64,118	-
packaging	350	126	115,593	43,740	-
packaging	625	448	775,442	92,130	-
packaging	420	282	456,163	112,237	-
packaging	3,420	1,222	574,775	99,767	-
packaging	388	245	340,800	68,907	-
packaging	559	384	730,667	129,683	-
packaging	367	232	200,000	32,649	-
packaging	264	134	168,331	22,488	-
JPMA	296	189	33,500	3,636	47
JPMA	591	378	50,000	7,272	40
JPMA	591	378	55,000	3,246	31
JPMA	296	189	35,000	17,790	123
JPMA	591	378	52,000	35,586	116
JPMA	296	189	56,000	2,088	58
JPMA	591	378	83,000	4,176	45
JPMA	296	189	35,000	-	-
JPMA	591	378	100,000	-	-
JPMA	96	61	46,000	1,518	142
JPMA	185	118	57,500	2,112	94
JPMA	222	142	44,000	3,360	72
JPMA	96	61	35,000	4,188	157
JPMA	185	118	39,000	7,164	112
JPMA	222	142	44,000	7,044	98
JPMA	296	189	30,000	5,400	53
JPMA	591	378	60,000	6,000	41
JPMA	591	378	70,000	2,100	35

Table 2.32 (cont.)

Table 2.32 (cont.)

JPMA	887	567	85,000	2,760	28
JPMA	1,183	756	95,000	3,600	24
JPMA	96	61	48,000	1,080	140
JPMA	96	61	32,000	2,520	123
JPMA	185	118	57,500	1,500	89
JPMA	185	118	37,500	5,100	93
JPMA	222	142	59,000	1,800	78
JPMA	222	142	39,000	5,760	84
JPMA	296	189	31,000	5,346	54
JPMA	591	378	54,000	14,622	61

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3 From thermodynamic efficiency to eco-efficiency

Reinout Heijungs
CML, Department Industrial Ecology, Leiden University, Netherlands

Abstract

According to Webster's Revised Unabridged Dictionary, efficiency is (1) the quality of being efficient or producing an effect or effects and (2) (in the context of mechanics) the ratio of useful work to energy expended. The first description obviously relates to more everyday language than the more science-focussed second description. This second one, however, is restricted to mechanics and thermodynamics. The efficiency concept is also used in economics, but there it appears to indicate the state of optimality, and not a quantifiable degree of optimality.

There is as yet no unambiguous and generally accepted definition of eco-efficiency. Obviously, eco-efficiency is a term that has emerged from everyday, rather than scientific arguments. It is, however, equally obvious that eco-efficiency should in the course of time, and in its development into a quantifiable and communicable term, be further specified on a scientific basis. Admittedly, consensus seems to be growing that an eco-efficiency indicator expresses the ratio between an environmental and a financial variable, witness the various texts by Schaltegger, the WBCSD, the OECD and the UN. But there is still much confusion.

This paper argues that this confusion may be due to an unconventional use of the term 'efficiency'. In order to develop a better understanding of the exact meaning of eco-efficiency, it reviews the thermodynamic origins of the efficiency concept. This serves as a point of departure for a generalisation of this concept to a form that will accommodate eco-efficiency as well.

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 79–103.

The interpretations of the term efficiency in the economic vocabulary are also reviewed in this context. I then present an axiomatic scheme for an efficiency indicator, on the basis of a ratio of input to output exergy. Its usefulness is illustrated by an example relating to iron production. Finally, the paper shows that the 'economy–environment ratio' – even though it is not an actual efficiency or eco-efficiency – is still a useful indicator. I propose to call this the 'eco-productivity'.

3.1 Introduction

Within the last decade, the term eco-efficiency has become popular in the realm of industrial ecology, cleaner production and related fields dealing with questions of sustainability. One would expect that such a popular term would have a clearly defined, unambiguous and generally agreed meaning. This is, however, not the case. As I will show, eco-efficiency can refer to a target, it can refer to an indicator, or it can refer to a tool. Quantitative indicators of eco-efficiency can differ in terms of meaning and/or units. As Schaltegger & Burritt (2000, p 49) remark:

> *In practice, the term has been given different meanings and, as a result, has little precision.*

This situation is clearly not conducive to the acceptance of sustainability as a field of scientific progress, while appropriate and universally accepted definitions of concepts such as eco-efficiency may well gain greater respect for the field.

In this paper, I explore some of the fields of study in which the term efficiency is used. From the overview thus obtained, I draw some conclusions as to the universal properties of the efficiency concept. This will enable me to provide a less ambiguous and more consistent definition of eco-efficiency.

3.2 The efficiency concept and term as used in a variety of fields

This section explores some of the places where the term efficiency shows up. I start with its use in everyday language, and end up in economics and statistics.

Everyday language

The term efficiency shows up in many everyday conversations, but its meaning is often blurred. People may speak of the efficiency of shopping only once a week, or the efficiency of using a certain computer program for managing personal finances. Obviously, this usage is based on a non-quantitative meaning of the term efficiency. Efficiency points to the state of being efficient, or being more efficient than another option. And efficient stands for practical, economical and time-saving.

Most everyday terms that enter the scientific vocabulary become more sharply and restrictively defined. Examples of such terms are force and resistance. Conversely, scientifically coined terms may acquire a broader meaning as they penetrate into everyday language. Examples of this process are energy and momentum. Efficiency is probably a term that belongs to the latter category. First defined in a scientific context, it has entered the common vocabulary. In doing so, it has not only widened its meaning, but has also shifted from a quantitative indicator of the degree of optimality to the state of optimality itself.

Philosophy

Philosophers have used the efficiency concept in a very different sense. Martin Heidegger (1977) discusses the concept of enframing, which offers a notion of revealing the essence of a thing or idea by thinking about that which shapes it, its causes (Beckman, 2000). Thus he distinguishes four causes (Anonymous, 2000):

- the *causa materialis* (the material): what the material is or is perceived to be, or by its use may become;
- the *causa formalis* (the form of the material): through the act of making, to an end, having a set form;
- the *causa finalis* (the end): a product, whether physical or derived from the material form and making;
- the *causa efficiens* (the makers making/the act): the doing or working out, being one in the same, to make, as an act or intention and to try to prove.

In the present context, the causa efficiens is of course what interests us most. It is a notion that goes back to Aristotle (Book II of his Physics) and is used to denote, for instance, the silversmith who brings about the effect that is the finished object, like the chalice in the famous example.

Thermodynamics

Triggered by recent developments in steam-driven engines, the French engineer S. Carnot (1824) analysed the quantitative aspects of energy conversion. His compatriot Clapeyron (1834) extended this analysis. A now famous concept is the Carnot engine (Adkins 1983), a system that is composed of two heat reservoirs at different but constant temperatures, between which a heat-absorbing device can move back and forth and deliver work, i.e. usable energy.

The efficiency of a Carnot engine is defined as the ratio of the net work delivered (W) to the net heat absorbed (Q). Traditionally, the Greek η (eta) is used to denote the efficiency, or more precisely, the thermal efficiency. The formula is

$$\eta = \frac{\Delta W}{\Delta Q}$$

Heat and work are energetic quantities, which may be expressed in a unit like joule (J). The efficiency is a ratio of two quantities with the same unit, and is thus a dimensionless pure number.

The theory of thermodynamics provides expressions for the efficiency. For a Carnot engine with the warmer heat reservoir kept at $T = T_h$ and the colder at $T = T_l$, the efficiency can be written as

$$\eta = 1 - \frac{T_l}{T_h}$$

This number is bounded between 0 and 1.[1] It approaches 1 when T_h is much larger than T_l. This explains, for instance, why electric power plants use low

[1] Occasionally, one may encounter an efficiency that is larger than 1 (or 100%), see e.g. Smith (2001). This then points to ambivalent definitions of the baseline energy level. For heating systems, an efficiency of 107% is often reported, since the condensation heat is included. With properly defined baseline levels, an energy efficiency can never exceed 100%, as this would violate the law of energy conservation and provide us with a perpetuum mobile.

temperature cooling water: it is a device that ensures a high energetic efficiency.

In fact, the expression for η is a maximum value, the Carnot efficiency, with which the actual efficiency of any Carnot engine may be compared. Thus, a theoretical maximum and an actual efficiency serve to distinguish irreducible and avoidable losses.

Engineering

Thermodynamics has a clear origin in engineering, but has grown into an independent academic discipline. The engineering sciences provide plentiful examples of the efficiency concept, in most cases still related to energy. In electric engineering, for instance, battery efficiency can be defined in two basic ways (Anonymous 2003):

- the amount of energy that can be drawn out of the battery divided by the amount of energy needed to charge the battery;
- the amount of energy actually drawn out of the battery divided by the total amount of energy stored in the battery.

The similarity to the original thermodynamic concept is obvious.

Ecology

Ecologists have defined various efficiency concepts (White et al. 1992). A well-known form is the Lindeman efficiency, which is defined as the ratio between energy flows at various trophic levels (Lindeman 1942). For instance, Slobodkin (1960, p 222) defines efficiency as

> *the energy per unit time taken from some population (the prey) as yield by some other population (the predator) to the energy per unit time ingested by the prey population.*

Just as in engineering and thermodynamics, efficiency is almost invariably defined in terms of energetic quantities.

Economics

In economics, the efficiency concept and term emerge in various contexts. However, almost all places where efficiency is mentioned or even defined appear to discuss the 'efficiency property', which denotes the quality of being efficient in the sense of being optimal. The term 'optimality' would be much better here. A few examples[2] are Pareto efficiency (which is indeed better referred to as Pareto optimality):

> *An allocation of resources is Pareto efficient if there is no way that any individual could be made better off without making some individual worse off;*

x-efficiency[3]:

> *X-efficiency is the effectiveness with which a given set of inputs are used to produce outputs. If a firm is producing the maximum output it can given the resources it employs, such as men and machinery, and the best technology available, it is said to be x-efficient.*

and Kaldor–Hicks efficiency:

> *Kaldor–Hicks efficiency (named for Nicholas Kaldor and John Hicks) is a type of economic efficiency that occurs only if the economic value of social resources is maximised.*

The classic textbook by Samuelson (1967, p 609) states the following:

> *Under perfectly perfect competition, [...] the resulting equilibrium has the efficiency property that 'you can't make any one man better off without hurting some other man.'*

Winch (1971) discusses efficiency in production and efficiency in exchange, always in the sense of a state that can be achieved when certain conditions are fulfilled. And finally (Schenck 2003):

> *An economy is said to be efficient if it operates on its production-possibilities frontier, and every point on this frontier satisfies the condition of production efficiency.*

We thus see clearly that efficiency in economics is not a quantitative property, but a state of affairs which applies or does not apply to the object of

[2] The first three examples are taken from http://en.wikipedia.org/wiki.
[3] X-efficiency is itself not clearly defined. Bannock et al. (1998, p 437) define it in the sense as above, as well as in the sense of the discrepancy between the best possible efficiency and the actually achieved efficiency. The latter form is also articulated by Schaltegger & Burritt (2000, p 50). Notice that the latter form assumes the definition of another type of efficiency.

investigation. Although it is sometimes possible to compare the degree of efficiency (‘Economy A is more efficient than economy B’), it is not possible to quantify it in either absolute (‘Economy A has an efficiency of 0.74’) or even relative (‘Economy A is 2 times as efficient as economy B’) terms.

Some other disciplines

In statistical inference, the concept of efficiency is used in a quantitative way. The power efficiency of a statistical test measures the sample size that is needed to achieve a certain discriminatory power (Siegel 1956). It is a measure relative to the most powerful test.

Sadi Carnot's father, L. Carnot, has been argued to have generalised the efficiency concept to a variety of fields; see Carnot (1984, p 416) for a treatment of military efficiency dating back to 1789.

There are many other fields where the term efficiency has permeated, e.g. logistics, pharmacology and psychology. I will not address all these fields, but in most instances, the everyday or economic interpretation of the word efficiency is meant: a state of optimality, or a non-quantitative measure of the deviation from such a state.

An intermediate conclusion

The examples of the use of the concept and term efficiency show two main uses:

- as a qualitative indication of an optimal state of a system;
- as a quantitative indicator of the closeness of the actual state of a system to the optimal state.

I inspect the situation in more detail below.

3.3 Towards a generalised efficiency concept

The survey presented in Section 3.2 shows that the efficiency concept is unclear: it is not defined in a precise way, and definitions differ quite substantially between different disciplines. Let us start with a linguistic analysis, parsing the usage of the term in sentences, and then proceed to provide a more precise definition.

Linguistic analysis

A review of the various efficiency concepts and definitions allows three conclusions to be drawn:

- Efficiency either indicates a state of optimality (like the Pareto efficiency) or a quantitative indicator of the closeness to optimality (like the Carnot efficiency).
- Both options require that a clearly defined state of optimality is described (for the Pareto case, this is a state in which no one can be better off without making another person worse off; for the Carnot case, it is a situation in which no energy is lost).
- The quantitative efficiency indicators have the property of being dimensionless pure numbers (like the ratio of the work supplied to the heat absorbed) which range between 0 and 1 (or 100%), and for which a value of 1 denotes the optimal or most efficient situation.

These three properties can be observed from a number of examples. In this section, they will serve as postulates in defining a generalised efficiency concept. The various examples also illustrate the distinctive use of four quite similar terms:

- the noun efficiency as used to refer to quantitative indicators (like 'the Carnot efficiency');
- the noun efficiency as used in compound nouns (like 'the efficiency indicator');
- the adjective efficient as used predicatively, i.e. to describe a state (like 'the economy is Pareto-efficient')
- the adjective efficient as used attributively, i.e. to characterise a noun (like 'the efficient economy').

A closer look at the above quotations reveals that some alleged definitions of the noun efficiency are in fact definitions of the adjective efficient (like that for Pareto efficiency). This implies that a precise definition of efficiency and/or efficient should be precise in the syntactic role that the term is supposed to play in a sentence. It also implies that one should take care in deciding whether to define nouns, adjectives, adverbs, or two or three of these.

Definitions and propositions

This section presents formal definitions of the terms efficiency and efficient, illustrates them, and derives a number of propositions.

Definition 1
The noun *efficiency* refers to the degree of optimality of a system. It can be a quantitative indicator, in which case it is a dimensionless pure number, measured on a ratio scale, bounded between 0 and 1, and with higher values signifying a higher degree of optimality. Alternatively, it can be a qualitative relative indicator, measured on an ordinal scale, and again with higher values signifying a higher degree of optimality.

Example 1
The Carnot efficiency and the statistical efficiency are examples of quantitative indicators of efficiency.

Example 2
In a comparison of firms, one can assert that 'firm A has a higher efficiency than firm B', without necessarily specifying the individual efficiencies of firms A and B (or even being able to so).

Definition 2
The adjective *efficient* refers to a system with a certain level or range of levels of optimality.

Example 3
A machine may be called efficient whenever its efficiency exceeds a predefined value, say 0.9.

Example 4
An allocation of resources is Pareto-efficient if there is no way that any individual could be made better off without making some other individual worse off.

Definition 3
The adjective *efficient* can be used as an adjective for systems that are efficient.

Example 5
A machine that is efficient may be called an efficient system.

Proposition 1
A general expression for the efficiency of a system is

$$\eta = \frac{U_{\text{out}}}{U_{\text{in}}}$$

where U_{out} is the amount of utility or useful goods produced by the system, U_{in} is the amount of utility or useful goods absorbed by the system, and numerator and denominator are measured in identical units to ensure that η is a pure number (cf. Le Goff *et al.* 1990).

Example 6
The Carnot efficiency and the Lindeman efficiency are examples of efficiencies that are in accordance with this proposition.

Proposition 2
The variables that measure the amount of utility or useful goods in numerator and denominator should be chosen in such a way that

$$U_{\text{out}} \leq U_{\text{in}}$$

or, alternatively,

$$U_{\text{in}} = U_{\text{out}} + U_{\text{loss}} \textit{ with } U_{\text{loss}} \geq 0$$

where U_{loss} is the non-negative amount of utility or useful goods lost in the operation of the system.

Example 7
The theory of non-equilibrium thermodynamics and dissipative systems (De Groot & Mazur 1984) provides an example of formulations in which an internal entropy variation (the ‘loss’ term) and an external variation (the difference between the ‘in’ and ‘out’ terms) are considered separately.

The utility indicator revisited: energy, entropy, exergy, quality?

The form in Proposition 1 is a ratio between two indicators of utility (or useful goods). This interpretation in terms of utility limits the freedom of selecting indicators in concrete cases. Moreover, the requirement in Proposition

2 produces additional constraints. This section explores some of the options left.

The thermodynamic origin of the efficiency concept and the requirement posed by Proposition 2, that there is a non-negative loss of utility by the operation of the system, induces a study of the notion of entropy in irreversible systems. Entropy is a quantity that characterises the state of a system. Loosely stated, it indicates the degree of disorder of the system. A system in equilibrium has maximum entropy (Adkins 1983).

An important property of entropy is that it cannot decrease when a system has no exchange of matter and/or energy with its surroundings. It can increase or remain constant. A system out of equilibrium can tend towards a state of equilibrium by producing entropy. On the other hand, a system in equilibrium will not move towards a non-equilibrium state, because this would mean diminishing its entropy. This rapidly leads to a discussion of irreversible changes. A spontaneous change from state A to state B that is associated with an increase of entropy cannot spontaneously proceed in the reverse direction, from B to A, because the entropy of state A is lower than that of state B.

In the context of ecological economics, the importance of entropy for economic systems had already been stressed well before the term ecological economics was coined. Nicholas Georgescu-Roegen (1971) insisted that economists had to consider what he called ‘the entropy law’. Ayres & Nair (1984) reinforced the importance of thermodynamics for economic processes, like manufacturing, and distinguished two types of efficiency: first-law efficiency ε, only taking into account the conservation of energy, and second-law efficiency η, also taking into account the irreversible production of entropy. They stress that the Carnot engine is a reversible process from a local perspective, but an irreversible process from a global perspective, as the system feeds on low entropy. Bianciardi *et al.* (1993) discussed the key difference between a reversible (or even irreversible) cycle and the economic process, which has no cycle character at all, but simply transforms inputs into outputs. Baumgärtner & de Swaan Arons (2003) extensively discussed this transformation process in the context of the necessity of generating waste and consuming energetic resources. The connection between the rate of entropy production, the maximum efficiency and the optimum efficiency has been made by Odum & Pinkerton (1955).

Ayres (1998) proposed to use exergy as a universal indicator of quality, which facilitates a comparison of resource and waste flows to and from a system. Exergy (see Rant 1956) is a quantity expressed in the units of energy, but representing solely the amount of available energy in relation to the surroundings. For instance, a material body with a temperature of 20°C has the capacity to perform work in a cold environment, but does not have this capacity in a warm environment. In the context of ecology, exergy has been introduced by Jørgensen (1992) as an indicator of the system's property. In this sense, it is supposed to capture the more holistic aspects, like the level of internal organisation (Jørgensen *et al.* 1995). Exergy can easily be linked to the concept of efficiency (Ayres 1998, p 194):

> *The more efficient (in the second law, or exergetic sense), the less exergy is embodied in the materials that must be discarded.*

Baumgärtner & de Swaan Arons (2003, p 117) observed that

> *Physicists usually prefer the entropy route, as entropy is the concept traditionally established in physics. On the other hand, exergy seems to be more popular with engineers and people interested in applied work.*

Another motivation for favouring exergy over entropy is that it is more easily linked to the discussion of the loss of something useful, as in the case of the Carnot engine (cf. Proposition 2). Using the jargon of entropy, Schrödinger (1994) artificially introduced negative entropy (sometimes referred to as negentropy) to be able to communicate in terms of dissipation.

More esoteric and less accepted indicators of utility are emergy and quality. Emergy is associated with the name of H.T. Odum (1983), and is a variable that aims to reflect embodied energy, in the sense of the energy that is needed in a cascade to produce a certain amount of energy. The principle of accounting for embodied amounts has been elaborated by others, like Scienceman (1987), who introduced emtropy, emformation, emprice, emonomics, emtelligence and empower. Links between emergy and more traditional indicators (exergy and entropy) have been made by Pillet *et al.* (1987). Finally, Funtowicz & Ravetz (1997) discussed the concepts of energy, entropy and exergy in relation to a fourth principle, that of quality. In this sense, quality is considered to be the appropriate term for reflexive systems, like those pertaining to life. No quantitative expression for such a quality indicator is available. This is probably an inherent property of this 'post-normal' concept, as quantifying quality sounds somewhat absurd.

Reviewing all these proposals to capture the notion of utility in quantitative terms, exergy is perhaps the most promising candidate, as it is defined relative to a reference situation, can be measured, and relies heavily on statistical mechanics and thermodynamics, and is thus firmly rooted in the microscopic and phenomenological theories of matter and motion. However, it should be clear from the above overview of utility indicators that no definite choice can be made at this moment.

Exergy as an indicator of utility in economics and environmental science

Exergy has been identified as a candidate for quantifying the utility aspect of numerator and denominator in expressions of eco-efficiency. Below, I briefly discuss a number of examples where exergy has been used as an indicator. The selection is by no means exhaustive, however; Göran Wall's website http://exergy.se/ gives a very long list.

Exergy has been introduced in economic or environmental accounting schemes by Ukidwe & Bakshi (2004), Hau & Bakshi (2004) and Sciubba (2001). The analysis of industrial, ecological and other systems by means of exergy has been discussed by Balocco et al. (2004), Chen (2005), Gong (2005), Jørgensen (1992), Silow & Oh (2004) and Szargut (2003). Links between exergy and the environmental life cycle assessment of products (LCA) have been described by Bakshi (2000), Cornelissen & Hirs (2002), Daniel & Rosen (2002), Dewulf & Langenhove (2002) and Finnveden & Östlund (1997). The use of exergy as a measure of ecological quality, pollution or environmental damage has been discussed by Bastianoni (1998), Connely & Koshland (1997), Fath & Cabezas (2004), Gong & Wall (2001), Rosen & Dincer (2001) and Seager & Theis (2002b). Overall contributions to the incorporation of exergy in industrial ecology have been made by Ayres (1998), Baumgärtner & de Swaan Arons (2003), Connely & Koshland (2001), Seager & Theis (2002a), Speigelman (2003) and Wall & Gong (2001).

It is obvious that the use of exergy in the analysis of industrial–ecological systems is not new. In fact, it has been argued by many proponents that exergy provides a natural instrument to address issues related to quality, loss of quality and damage. However, and this is the key point in which the present contribution differs from previous studies, most texts have focussed on the development of indicators of resource quality or pollution or the use of such indicators for optimisation purposes. The use of the ratio of two exergy indicators to form an efficiency indicator is a novel aspect. Only Bastianoni & Marchettini (1997) have provided a ratio of two exergy (or emergy) terms, albeit in a reciprocal way: the cost of production per unit of organisation.

3.4 Eco-efficiency, or the efficiency concept in industrial ecology

This section first reviews some of the definitions of the term eco-efficiency and its origin, and then proceeds to propose an alternative definition on the basis of the considerations presented in the previous sections.

The historical setting

The World Business Council for Sustainable Development (WBCSD) is often quoted as having introduced the concept of eco-efficiency (see, e.g., Saling 2002). On its website, WBSCD (2004*a*) itself states that:

> *The basic business contribution to sustainable development is eco-efficiency, a term the WBCSD invented in 1992.*

Indeed, Schmidheiny (1992, p 10) wrote in the (W)BSCD book *Changing Course* that:

> *Industry is moving toward 'demanufacturing' and 'remanufacturing' – that is, recycling the materials in their products and thus limiting the use of raw materials and of energy to convert those raw materials [...] It is the more competitive and successful companies that are at the forefront of what we call 'eco-efficiency'.*

As this was a non-technical book, it did not define this concept in precise terms. In fact, it uses the term somewhat loosely, as can be seen from the following quotation from Schmidheiny (1992, p 98):

> *[...] growing number of companies are regularly raising their 'environmental efficiency' – the ratio of resource inputs and waste outputs to final product.*

This suggests a definition (resource inputs and waste outputs divided by final product), but at the same time, companies seem to be proud about increasing this ratio instead of decreasing it.

Schaltegger (1997) corrects the historical dating by pointing out that a 1990 paper by Schaltegger & Sturm, in German, introduced *ökologische Effizienz*, which can be translated as *ecological efficiency*. To them, eco-efficiency is the ratio between the intended effects and other positive external effects and the damage created:

$$\eta = \frac{\sum \text{intentend effects} + \text{postive external effects}}{\text{damage created}}$$

Both numerator and denominator are expressed in SE, which stands for *Schadschöpfungseinheiten* (damage creation units; a hypothetical overall measure of environmental damage). The eco-efficiency according to Schaltegger & Sturm (1990) is hence a dimensionless pure number. They give an example of two hypothetical filters with eco-efficiencies of 3 and 1.7. Note that the ratio can exceed 1.

The panoply of definitions

Eco-efficiency has entered the scientific and managerial literature in many forms. The Word Spy website[4] presents a lemma on 'eco-efficiency', defining it as

> *The ability to manufacture goods efficiently and with as little effect on the environment as possible.*

In this definition, eco-efficiency is described as something that can be achieved, in other words, it is not a measure of optimality but optimality itself. This can be contrasted with another quotation on the WBCSD's website (WBCSD 2004b):

> *Eco-efficiency is a management strategy that links financial and environmental performance to create more value with less ecological impact.*

Here efficiency is described as a strategy used to arrive at a state of optimality. According to OECD (1998, p 69), opinions differ as to whether eco-efficiency should be defined in a tight indicator-type sense, or in broad way,

> *allowing many stakeholders with differing interests to sign up in strategies for eco-efficiency.*

[4] See http://www.wordspy.com/. This is a web service that 'is devoted to *lexpionage*, the sleuthing of new words and phrases. These aren't "stunt words" or "sniglets," but new terms that have appeared multiple times in newspapers, magazines, books, Web sites, and other recorded sources.'

The use of eco-efficiency as a quantitative indicator has been mentioned many times. The first instance is that by Schaltegger & Sturm (1990), which was given in the previous section. In addition, there are the definitions proposed by the organisers of the conference to which this paper was delivered (Anonymous 2004a):

> *Eco-efficiency may be reserved for the ratio between economy and environment, with environment in the denominator;*

the one by Saling et al. (2002, p 203):

> *Eco-efficiency expresses the ratio of economic creation to ecological destruction;*

and the official UN definition (Anonymous 2004b, p 1):

> *An eco-efficiency indicator is the ratio between an environmental and a financial variable. It measures the environmental performance of an enterprise with respect to its financial performance.*

Similar, but sometimes different, usage of the term efficiency can be found in many other places. In any case, the form of the ratio between an economic and an environmental quantity is often seen (OECD 1998) and appears to have become the standard way of thinking on eco-efficiency. The reader should be cautious, however, about detecting other forms. A very recent paper (Gössling et al. 2005, p 418) in a renowned journal (Ecological Economics) uses eco-efficiency as

> *the ratio of CO_2-e (kg) to turnover (€)*

precisely the opposite of the definitions quoted above.

It is clear that in the quantitative definitions of eco-efficiency, it is not a dimensionless pure number[5], and it has no theoretical limit, like the magic limiting value of 1 for a Carnot engine. Even though it may be a useful indicator, it is not something that most scientists would consider to represent a measure of efficiency.

[5] Schaltegger & Burritt (2000, p.50) state that 'efficiency is a multi-dimensional concept because the units in which input and output are measured can vary'. As long as the numerator and denominator vary in the same way, I agree, although I prefer to replace the term 'multi-dimensional' by 'dimensionless pure number'. I do not agree when numerator and denominator are allowed to be different.

A thermodynamics-inspired definition

Armed with the knowledge from different fields of learning, we are now in a position to reformulate the concept of eco-efficiency in a scientific way, making it a dimensionless pure number with a theoretical limiting value of 1.

Definition 4
The noun *eco-efficiency* refers to the degree of optimality[6] of a system that comprises economic activity and ecological substrate. It can be a quantitative indicator, in which case it is a pure number, measured on a ratio scale, bounded between 0 and 1, and with higher values signifying a higher degree of optimality. Alternatively, it can be a qualitative relative indicator, measured on an ordinal scale, again with higher values signifying a higher degree of optimality.

Proposition 3
A general expression for the eco-efficiency of a system is

$$\eta = \frac{B_{\text{out}}}{B_{\text{in}}}$$

where B_{out} is the exergy of the outputs of the system and B_{in} is the exergy of the inputs of the system.

Thermodynamic analysis (Baumgärtner & de Swaan Arons 2003, pp 118–119) demonstrates that this ratio will never exceed 1, that for physico-chemical reasons it often has a theoretical maximum that is less than 1, and that in industrial practice it will be even less, especially when man's impatience forces processes to proceed far from equilibrium, so that finite-time thermodynamics takes over (Andresen 1990). Phrased in this way, eco-efficiency can be disentangled into a part due to inevitable conversions and a part due to sub-optimal operation of industrial systems (Baumgärtner & de Swaan Arons 2003), in the same way as the Carnot efficiency provides an upper bound that reflects inevitable losses, while the actual thermodynamic efficiency reflects sub-optimal features as well. In our view, this makes

[6] It should be recognised that a term like 'optimal' or 'optimality' seems to imply a favourable attitude. Although eco-efficiency (or a high level of eco-efficiency) is indeed a target for most practitioners, the definition given here is intended to be more neutral, so that it even applies to more critical stances (Hukkinen 2001, 2003).

eco-efficiency an indicator whose value surpasses that of a mere indicator of wealth per unit of damage.

The expression for eco-efficiency can be rewritten as follows:

$$\eta=\frac{B_{\text{out}}}{B_{\text{in}}}=\frac{B_{\text{in}}-B_{\text{loss}}}{B_{\text{in}}}=1-\frac{B_{\text{loss}}}{B_{\text{in}}}$$

thus stressing both the confinement between 0 and 1, and the psychological aspect of accounting for losses.

It should be noted that our definition of eco-efficiency primarily deals with a system, or one building block of a system. In considering a cascade of building blocks, one might be interested in analysing a cumulative exergy ratio. From the viewpoint of one of these building blocks, this would then imply an analysis of embodied exergy. It remains to be investigated to what extent this corresponds with the emergy/exergy ratio proposed by Bastianoni & Marchinetti (1997) to characterise the sustainability of organisms and other self-organising systems. Likewise, the sustainability indicator based on a ratio of emergies (Bastianoni & Marchinetti 1996) provides an interesting alternative.

An example

To illustrate the exergy-based efficiency concept, let us regard the example of the production of pure iron from iron ore, discussed by Baumgärtner & de Swaan Arons (2003). The theoretically optimal reaction equation is

$$2\text{Fe}_2\text{O}_3+3.76\text{C}+0.76\text{O}_2\rightarrow 4\text{Fe}+3.76\text{CO}_2$$

The useful exergy produced is 1505.6 kJ, whereas the exergy used up is 1578.7 kJ. The eco-efficiency is thus

$$\eta=0.95$$

The real production process can be written as

$$2\text{Fe}_2\text{O}_3+12.42\text{C}+9.42\text{O}_2\rightarrow 4\text{Fe}+12.42\text{CO}_2+\text{heat}$$

with an exergy input of 5166.6 kJ, and an eco-efficiency of

$$\eta=0.29$$

hence far below the theoretical equivalent of the Carnot efficiency of 0.95. Clever process redesign can now easily focus on those processes with a large gap between maximum and actual eco-efficiency.

The advantage of the dimensionless form of eco-efficiency with a limiting value is evident: it is immediately obvious how far off 0.29 is from an unreachable 1.0, but also from a theoretically limiting 0.95.

Eco-efficiency and eco-productivity

A final question concerns the interpretation of the ratio of economy to environment, which is the operationalisation of eco-efficiency according to many other authors (see Section 'The panoply of definitions' above and many contributions to the present volume). We suggest that the term 'eco-productivity' captures this ratio much better than the term 'eco-efficiency'. In general, economists define productivity as (see Bannock et al. 1998, p 334)

> *the relationship between the output of goods and services and the inputs of resources used to produce them.*

This concept appears in a variety of specific forms, like labour productivity and capital productivity. In general, factor productivity denotes the output of goods and services per unit of factor production used up. As the environment (including natural resources) is considered to be a factor of production by many (ecological) economists, eco-productivity would simply mean the output of goods and services per unit of environment that is used up.

The above form of eco-productivity coincides very well with the E/E logo of the International Conference on Quantified Eco-Efficiency Analysis for Sustainability (shown in this volume), provided that the first E means Economy and the second E means Environment. But the logo applies equally well to this paper's interpretation of eco-efficiency, as E/E, where both E's stand for Exergy. As a matter of fact, the completely monetarised form of ecological damage fits into this picture as well, if the logo is changed to €/€. In any case, the mere use of the metaphorical E/E proves to be too ambiguous to serve as a genuine starting point.

Can the economy-to-environment ratio, formerly referred to by some as eco-efficiency, and here proposed as eco-productivity, benefit from the thermodynamic considerations provided in this paper? I think it cannot. The numerator, economic performance, is a clearly market-driven parameter. If torn jeans are more valuable on the market than spotless ones, the thermodynamic notions of entropy and exergy cannot help. It is man's capricious preferences which determine the demand function, and thereby codetermine prices. For the denominator, the situation is slightly better, but not much. Exergy can provide a measure of pollution and resource quality, but it is not the whole story. Moreover, for monitoring and benchmarking reasons, companies and policy officials often wish to concentrate on partial aspects of the environmental performance. Thus, the eco-productivity is often expressed in €/kg CO_2, €/kg resource, or other partial measures of environmental factor input.

3.5 Conclusion

The above review of the meaning and interpretation of the term efficiency has provided a basis for defining a clear and unambiguous eco-efficiency indicator, based upon thermodynamics, and accounting for differences in quality. This indicator has the nice property of being a dimensionless pure number between 0 and 1, where 1 is the theoretical limit value in the case of optimal operation. The basis for this definition, exergy, ensures the intuitive property that losses are indeed interpreted as a loss of quality. It opens the way for quantitatively expressing and analysing fundamental (i.e., insurmountable) and practical (i.e., surmountable) inefficiencies in industrial operation. In addition, the indicator that measures the ratio of economic value to environmental damage, and that has become known under the name of eco-efficiency, has been shown to be an indicator like labour productivity and capital productivity, and can therefore better be renamed eco-productivity.

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4 The price of toxicity. Methodology for the assessment of shadow prices for human toxicity, ecotoxicity and abiotic depletion

Toon van Harmelen, René Korenromp[a], Ceiloi van Deutekom[b],
Tom Ligthart, Saskia van Leeuwen[c] and René van Gijlswijk
TNO Institute of Environment, Energy and Process Innovation, Apeldoorn, The Netherlands
[a] *now working at Adecs Oost, Zwolle*
[b] *now working at Yacht*
[c] *now working at ABK InnoVent, Doetinchem*

Abstract

Weighting of environmental impacts is necessary to arrive at a single environmental indicator. One of the methods to weigh impacts, which has been operationalised for a number of impact categories in the Netherlands, is known as the shadow price method, using the highest acceptable costs for mitigation measures as a weighting factor. Up to now, no shadow prices were available for the more complex and less documented Environmental Impact Categories (EICs) in the field of human toxicity, ecotoxicity and depletion of abiotic materials. Therefore, a method was developed and applied to assess the shadow prices of these EICs. It consists of four steps: (1) characterising current environmental policy; (2) concentrating on the most relevant substances; (3) collecting abatement cost data and (4) calculating the shadow price.

The paper describes the method, discusses the results and concludes by presenting the full set of shadow prices in the Netherlands for the ten EICs of the CML-2 method. They are ready to be applied in the assessment of environmental profiles and the evaluation of measures in cost–benefit analysis, according to present policy preferences. We show that the external

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costs of the toxicity impact categories are, on average, substantial compared to those of the EICs for which shadow prices had already been established.

4.1 Introduction

The lack of comparability of environmental impacts poses a problem to investors, designers and not least to environmental policy-makers. it is hard to decide which appliance is more environmentally friendly: the ozone-depleting high-efficiency fridge or the ozone-friendly but more power-hungry fridge. In such cases, environmental impacts need to be weighed. One of the methods to do this is known as the shadow price method. It uses the highest acceptable costs for mitigation measures as a weighting factor and has been operationalised for a number of impact categories in the Netherlands. For instance, the Dutch Ministry of Public Works uses the shadow price method in combination with the life cycle assessment method called CML-2 (introduced by the Leiden University Institute of Environmental Sciences)(Guinée et al. 2001) in their life cycle impact assessment model DuboCalc to calculate the environmental impact of infrastructure works (Davidson and Wit 2003). The advantage of using shadow prices is that different environmental impacts are translated into external costs that can be compared with each other and with the internal production costs. The danger, of course, is that certain intrinsic values are underappreciated and get lost in the total cost analysis.

Several sets of shadow prices have been assessed, mainly for near-future targets of well-documented Environmental Impact Categories (EIC) such as climate change, acidification, ozone depletion, tropospheric ozone formation and eutrophication, e.g. by the organisations CE0, NIBE (Twin) and TME (KPMG Sustainability and CE 2002; NIBE Research 2002; Jantzen 2002). Internationally, the shadow price of CO_2 is also often referred to as the price of CO_2 on the emission trading market or the marginal reduction costs of national climate policies. However, shadow prices have so far not been available for the depletion of abiotic materials (ADP) and the toxicity-related categories (human toxicity potential – HTP, marine aquatic and sediment ecotoxicity potential – MAETP and MSETP, fresh-water aquatic and sediment ecotoxicity potential – FAETP and FSETP, or terrestrial ecotoxicity Potential – TETP). ADP relates to natural resources such as metals and fossil and nuclear fuels, the others cover pollutants including metals, pesticides, polycyclic aromatic hydrocarbons (PAHs), non-aromatic organic substances and inorganic substances.

We have developed a method to assess the actual shadow prices for these complex impact categories. This paper presents the methodology to assess the shadow prices of the human toxicity and ecotoxicity (from now on shortly referred to as toxicity) and abiotic depletion, as well as the resulting set of shadow prices.

4.2 Methodology

Alternative approaches

Different methods can be used to assess societal preferences for environmental quality as a basis for weighting or prioritising environmental impacts. Hofstetter (1998) and Huppes et al. (2002/2003) made a distinction between *who* decides on the priority and *how* the priority or preference is assessed.

As to the decision-makers, they found the government representing society most relevant for applications in a policy context. As to the way preferences are assessed, they distinguished between preferences directly assessed by statements and preferences indirectly revealed by actual, observed behaviour or market-based information. In addition to this, we – like Vogtländer– think it is important to make a distinction between weighting and valuing or monetisation methods; the former use points or percentages to weigh environmental impacts, while the latter use the value in monetary terms to assess the importance of these impacts. In our view, it is an advantage to express preferences in euros, to allow measures to be prioritised in relation to production costs and other economic activities. This makes the comparison explicit, although one should evaluate qualitative differences as well. In fact, the analysis should be a basis for a discussion of priorities, rather than provide a black box answer. Table 4.1 presents an overview of methods to assess societal preferences, using this terminology.

Table 4.1 Overview of four basic approaches to weighting (%) and valuation (€) methods to assess societal preferences for environmental quality, after Huppes et al. (2002/2003) and Pearce (2000)

How / **Who**	**Stated** Panels, hypothetical markets (future)	**Revealed** Conventional or surrogate markets (present)
Collective policy	**Stated collective preferences** Prevention costs/targets marginal costs [€] Distance-to-target [%]	**Revealed collective preferences** Prevention costs/actual marginal costs [€] [a] Replacement costs or damage costs [€]
Private person	**Stated private preferences** Contingent valuation (€, willingness-to-pay, willingness-to-accept compensation) Discrete choice modelling [€] Conjoint analysis [%]	**Revealed private preferences** Hedonic pricing [€] Travel cost [€]

[a] Used in this paper to monetise toxicity.

Stated collective preference methods include two main categories, viz. prevention costs methods and distance-to-target methods. In a distance-to-target method, the weighting factors are deduced from environmental policy targets with respect to emissions or concentrations. The ratio between the stated future target and the present level gives the weighting factor. Prevention costs methods or averting behaviour or avoidance costs methods derive the preference from the marginal costs of meeting emission reduction targets and infer preferences from actually observed market-based information. This method has been operationalised for the Netherlands by CE under the name of 'shadow price method' (Wit et al. 1997).

This method can also be implemented for the actual marginal costs resulting from present environmental policy measures and regulations. It does not involve stated preferences but represents *revealed collective preferences*, since the market is not hypothetical but real. Although theoretically different from the stated collective preference method as implemented by CE, we do not expect the difference to be very large for short- and medium-term policy targets. The deviation from a policy target might fall within the uncertainty interval of the estimated shadow price.

Another method to monetise the revealed collective preferences is using replacement costs or damage costs, where the costs incurred to replace or repair e.g. damaged health, crops and buildings is used as the value of the environmental impact. This method is difficult to use for toxicity, where the impacts on e.g. health or ecosystems are difficult to quantify and repair.

Stated private preferences elicit preferences directly with the help of questionnaires for panels. Of the methods in this category, contingent valuation expresses preferences directly in monetary terms, whereas conjoint analysis is based upon ranking. Discrete choice modelling is a mixture of the two methods, in which the value of environmental quality is inferred from the accepted cost difference between two goods that differ in terms of one environmental quality aspect.

Revealed private preferences include hedonic pricing, where the influence of environmental factors such as noise on the market prices of e.g. houses is used as the value of the environmental impact, and the travel cost method, in which the price a consumer is willing to pay for a visit to a site (e.g. a recreational site) is regarded as the value of the environmental impact. The latter method in particular is very limited in terms of the environmental aspects included.

In the present paper, we monetise the revealed collective preferences with respect to toxicity and depletion of abiotic materials by means of the avoidance costs resulting from present policy regulations, as highlighted in black in Table 4.1. Besides our preference for monetary values, another reason for using actual marginal costs is that cost data on mitigation measures to meet future emission objectives are hardly available. In fact, toxicity policy and mitigation options and costs have not been elaborated to the same extent as those of e.g. acidification and climate change, where there is a long history of intense international research, policymaking and negotiations on single national equivalent emission reduction targets.

The concept of shadow prices

There is a demand for environmental quality or damage limitation on a virtual market for environmental quality. In this market, the willingness to pay a high price increases with the emission level of pollution, and a supply of emission mitigation measures is available that cost more per unit of reduction at higher reduction levels. If this market existed, an equilibrium price would arise at the intersection of demand and supply, as illustrated in Figure 4.1.

Since the environmental market is a virtual market and environmental costs are so-called external costs, the government has to set an emission target to improve the environmental quality. The price level at the intersection between the emission objective and the supply of available emission mitigation is called the shadow price, being the highest price paid by society to improve environmental quality that is still acceptable to the government. The shadow price is the extent to which total costs change as a result of a change in a limiting factor, in this case an emission objective.
The total environmental costs to society will be the costs of mitigation (the shaded area under the supply curve) plus the damage to the environment, being the remaining emissions multiplied by the price level that society is willing to pay (according to the demand curve). In market equilibrium, this is the equilibrium price.

The government will aim its emission objective at the intersection of demand and supply, since at this point the virtual environment market is in equilibrium according to society. This is known as the societal optimum. Under the assumption that the government manages to design a policy whose shadow price equals the equilibrium price, the shadow price multiplied by the remaining emissions indicates the environmental damage as perceived in (and accepted by) society. This principle is used when applying the shadow price method.

Methodology for the assessment of shadow prices

We have developed a new method to assess the shadow prices of present policy regulations. For the well-known EICs such as acidification and climate change, an inventory of mitigation measures and costs derived from policy plans sufficed to assess the marginal costs of national abatement policies for a certain Environmental Impact Category. For these cases, national emission reduction cost curves and single national equivalent emission reduction targets are available, which makes the assessment of a shadow price fairly straightforward. For toxicity, the information is less

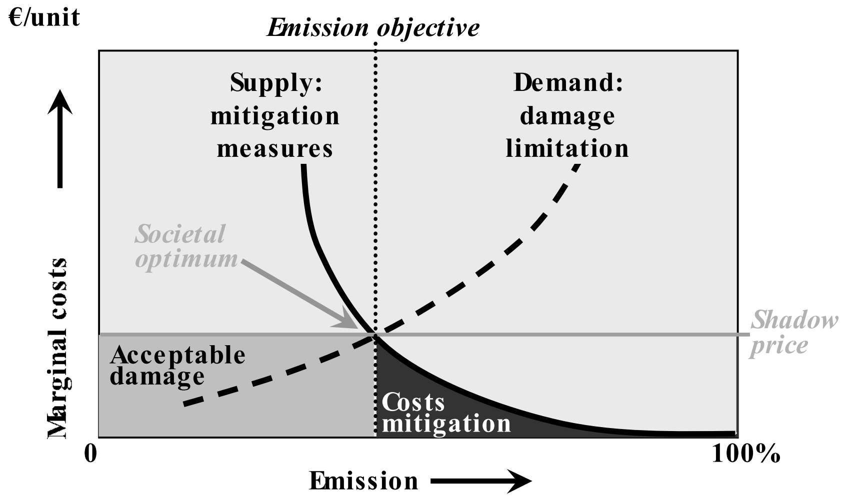

Figure 4.1 In a virtual market, demand for environmental damage limitation and supply of emission mitigation by measures will result in an equilibrium price for environmental quality. If a government's emission objective crosses the equilibrium point, the shadow price is optimal and equal to the equilibrium price

clear, and no single national equivalent emission objective is available. Furthermore, the number of relevant pollutants and the data quantities are so huge that a structured approach is required. The assessment method consists of four steps:

- o *Characterisation* of current environmental policy for each impact category.
- o *Concentrating* on the most relevant substances per impact category.
- o *Collection* of abatement cost data by means of literature research and interviews.
- o *Calculation* of the shadow price based on the cost-effectiveness of abatement measures.

1. Characterisation

Since the present environmental policies and regulations determine the mitigation measures that have been taken so far, the first obvious step is to look at the areas for which regulations are currently in place with respect to the EICs under consideration. This provides the necessary context for interpreting and understanding the data collected in the next steps.

2. Concentration

We need to concentrate on a selection of relevant substances for each impact category, since the number of pollutants in combination with initial media is already exceeding 200 items. Therefore, we calculated the emissions in 1,4-dichlorobenzene equivalents (1,4-DCB) using CML-2 characterisation factors (Guinée et al. 2001) and selected the most important pollutants for each EIC on the basis of three criteria:

1. share in national and sector equivalent emission in 1990;
2. historic change in equivalent emission over the 1990–2000 period;
3. present policy pressure to take measures.

The year 1990 was selected to ensure pollutants that have been greatly reduced are still included in the selection, since these pollutants are very important for the assessment of the shadow price.

We used a target group analysis of data from the Environmental Pollutant Emission Register (EPER) for the Netherlands to assess which company, process or other emission source is responsible for significant reductions. This allowed us to focus and increase the effectiveness of our data collection, and to assess which important sources have been covered and which have not (yet).

3. Collection

Data were collected firstly by telephone interviews with selected companies. Although we used a very specific approach in terms of questions on reduction of pollutants, little information was collected. Hence, most of the information on measures currently available for mitigation in the EIC under consideration was collected from the relevant national and international literature.

4. Calculation

A large number of measures are available to reduce emissions of one or more pollutants. To calculate the marginal costs, the additional costs of each measure are simply divided by the additional equivalent emission reduction of all pollutants. If pollutants contribute to more than one toxicity category, toxicity impacts need to be weighed to calculate the marginal costs. Although toxicity is expressed in 1,4-DCB equivalents in all EICs, its meaning may differ between EICs. For instance, effects in humans are not directly comparable with ecotoxicity in marine waters. We therefore developed a cost allocation method, consisting of two steps, (1) weighting the environmental impact categories; and (2) allocating costs by relative contribution to the environmental impact.

The initial weight of the EIC can be varied to assess the sensitivity of the results to the assumed weight. To avoid weights being arbitrarily chosen, several so-called policy perspectives were developed to characterise the relative importance of EICs. Furthermore, an iterative procedure was established to weigh the EICs with the resulting shadow costs.

Finally, the calculated shadow prices are compared with the actual environmental expenditures reported in the annual national environment report (*Milieubalans*) (RIVM 2000).

4.3 Assessment of shadow prices

Characterisation of current environmental policy

Toxicity is an important EIC in Dutch environmental policy. The annual expenditures are € 1.8 billion (year 2000; source RIVM 2000), which is more than the expenditures on climate change, acidification and eutrophication combined. Quantitative targets have been set for different compartments, either in terms of concentration limits or emission reduction targets for industrial sectors. The latter are voluntary agreements (VROM 2001).

For emissions to air, explicit concentration limits have been set for many organic pollutants and heavy metals, especially for combustion processes (PAHs, volatile organic compounds or VOC, dioxins and PM_{10} i.e. particulate matter with diameters up to 10 μm). Many pesticides have been banned or have a maximum allowable concentration (MAC) value.

For emissions of toxic compounds to water, emission standards have been set for waste water discharged by companies. Furthermore, sewage treatment plants have increasingly strict concentration limits for VOCs and heavy metals.

The policy on soil has changed over the past decades. It started in the 1960s and 1970s with a campaign of rigorous soil sanitation, which has been successful. The present view is that maintaining this sanitation standard is too expensive, and that soils should only be decontaminated if this is cost-effective. This means that the highest acceptable costs, i.e. the marginal costs, are 0. Nevertheless, target MAC values have been established for heavy metals, and the use of building materials has been regulated in the context of building materials regulations (*Bouwstoffenbesluit*).

Depletion of abiotic materials

As regards depletion of abiotic materials, only an indicative long-term target value has been set. No objective has been formulated and the accompanying policy has not yet been developed – only an indicator is under development. Up to now, quantitative targets have been set only in waste and energy policies. The current waste policy aims to avoid dumping waste at landfills, so recycling and waste incineration with energy recovery are equally prioritised. This means that materials depletion is not the single basic goal of this policy. Energy conservation is promoted for a number of reasons, including decreasing fuel import dependency, increasing supply security, reducing greenhouse gas emissions and avoiding resource depletion. Although it is very hard to distinguish between the various goals, resource depletion does play a role. However, in terms of antimony equivalents defined according to economic reserves depletion, as in CML-2 (Guinée 2001), consumption of fossil fuels accounts for less than 1% of the total equivalent materials depletion. It is not possible to attach a shadow price to a material that has such a low priority in equivalent terms, which illustrates that the equivalent method is not consistent with the present policy as regards energy resources.

For all other materials, no quantitative limits are being enforced, which means that market prices reflect economic scarcity.

Other environmental impacts that are not reflected in the market prices are included in the other EICs. These should not be included in the shadow price of abiotic depletion, to avoid double counting. It is therefore concluded that the best estimate of the present shadow price for depletion of abiotic materials is € 0 per Sb equivalent. In the rest of this paper, attention will be focused on the EICs involving toxicity.

Concentration on relevant substances and sectors

Table 4.2 presents the 95% percentiles of 1,4-DCB equivalent emissions of pollutant-initial medium combinations for the various EICs. To calculate the equivalent emissions, the characterisation factor for each pollutant-initial medium combination was combined with data from the Pollutant Emission Register for the Netherlands. Focusing on the 95% percentile for each EIC resulted in a reduction from over 200 to 30 pollutant-initial medium combinations. In fact, only 10 so-called priority pollutants determine the 95% percentile of equivalent emissions for each EIC. The emissions are presented for the year 1990 to ensure that pollutants that have recently been substantially reduced are still included in the selection. The reduction of the priority pollutants, presented over the period 1990–2000, is a crucial indicator of the mitigation measures that have recently been taken.

The initial medium determines the characterisation factor used, since the USES model calculates these factors on the basis of dispersion and exposure routes in combination with the toxicity of a pollutant (Guinée 2001). This also explains why an emission to air can result in marine ecotoxic effects.

Table 4.2 clearly shows that FAETP and FSETP are dominated by emissions to fresh water, TETP by emissions to soil and to a lesser extent to air, and HTP is largely determined by emissions to air. Note that MAETP and MSETP are dominated by emissions to air. This is caused by high characterisation values due to very long residence times.

The selected priority pollutants are a few heavy metals (relevant for all EICs), PAHs (for all EICs except TETP) and VOCs, benzene, dioxin and ethylene oxide (only important for HTP).

The equivalent emission reductions for each EIC over the 1990–2000 period are presented in the bottom row of Table 4.2. Equivalent emissions have decreased for all EICs, although terrestrial and fresh water EICs, with a reduction of one fifth to a half, are lagging behind reductions for HTP (more than halved) and marine EICs, with a reduction of three quarters.

Table 4.2 Contributions of priority pollutants to each individual EIC (95% percentile of 1,4-dichlorobenzene equivalent emissions), presented as emission shares to each EIC for 1990, and the total and reduction over the 1990–2000 period

Pollutant	Initial medium[a]	HTP eq	FAETP eq	MAETP eq	FSET P eq	MSET P eq	TETP eq	Reduction 1990-2000
Chromium (III)	Agr. soil						3%	**100%**
Copper (II)	Agr. soil		11%	2%	14%	3%	1%	22%
Mercury (II)	Agr. soil						1%	**100%**
Zinc (II)	Agr. soil		2%		2%		5%	2%
Arsenic	Ind. soil						2%	-50%
Chromium (III)	Ind. soil						***49%***	-26%
Copper (II)	Ind. soil				1%			-79%
Lead (II)	Ind. soil						1%	0%
Nickel	Ind. soil		1%		1%	1%		-10%
Acrolein	Air		10%		4%		2%	***30%***
Benzene	Air	19%						***45%***
Beryllium	Air			1%				**98%**
Chromium (III)	Air						4%	59%
Chromium (VI)	Air	1%						**97%**
Dioxins	Air	1%						**95%**
Ethylene oxide	Air	3%						**93%**
Hydr. fluoride [b]	Air			17%		7%		***47%***
Mercury (II)	Air						10%	**80%**
Nickel	Air	3%	1%	7%	1%	8%	1%	64%
Nitrogen oxides	Air	1%						20%
PAH (6 Borneff)	Air	66%						55%
Vanadium	Air	2%	9%	69%	12%	**77%**	20%	**89%**
Acrolein	Freshwa.		***30%***		11%			-2%
Benzo[a]pyrene	Freshwa.		14%		21%			74%
Fluoranthrene	Freshwa.		3%		5%			54%
Copper (II)	Freshwa.		3%	1%	4%	1%		***31%***
Nickel	Freshwa.		3%	2%	5%	3%		***34%***
PAH (6 Borneff)	Freshwa.	5%	11%		17%			64%
Zinc (II)	Freshwa.		1%		2%			22%
Nickel	Mar.wa.			1%		1%		***34%***
Total		100%	100%	100%	100%	100%	100%	77%
Reduction 1990–2000		56%	***35%***	**76%**	***47%***	**79%**	21%	

Shares of 25%-50% bold+italic; 50%-75% shaded; > 75% bold + shaded.
[a] Ind. = Industrial; Agr. = Agricultural; wa. = water; Mar. = Marine.
[b] The CML2 characterisation factors for hydrogen fluoride have been decreased by a factor 80 to correct for the incorrectly assumed long residence time.

Obviously, a large number of mitigation measures have been implemented over the last decade. This is confirmed by the rightmost column in Table 4.2, which presents the equivalent emission reduction per pollutant-initial medium combination for the same period. Note that the emission reduction rate due to a measure is equal for each EIC affected; only relative contributions can differ for different EICs. Equivalent emissions to soil have not been reduced according to the EPER register. The 100% reduction in chromium and copper to agricultural soil seems more likely to be a result of a monitoring error than of a strict mitigation measure.

In the further analysis, priority is given to heavy metals (all compartments), PAHs (air and water), dioxin, hydrogen fluoride and organic compounds (air). In an additional analysis, important target groups and companies were identified to focus our data collection. For practical reasons, the results are not presented here, and the reader is referred to the project report (Harmelen et al. 2003).

Collection of marginal cost data on measures

In the previous step, a number of pollutants, sectors and companies were selected. This allowed us to interview approximately 50 companies by telephone, asking questions tailored to their specific situation in terms of emission reductions of pollutant X. In addition, we used the national (Buurma et al. 2000; Dellink and van der Woerd 1997; Vogtländer 2001; Wagemaker et al. 1999) and international (Peirce et al. 2002) literature on emission reduction cost curves of specific pollutants. However, international data in particular cannot be exactly fitted to the Dutch situation, since the composition of the cost curve in terms of reduction potentials may differ and the position of the emission reduction objective in the cost curve is not clear. Nevertheless, it is possible to use them as a first estimation, since marginal cost curves generally consist of a flat part and a steep part. We selected the flat part, offering the greatest potential at relatively low marginal costs. Hence, we assumed that the Dutch government has a policy that forces companies to implement the most cost-effective measures in terms of euros per 1,4-DCB equivalent avoided. If fewer measures are implemented, the prices are not very different, since this part of the cost curve is flat. If more measures are currently implemented, prices are many times higher, since these measures fall in the steep part of the cost curve, so little additional reduction is reached at high additional costs.

For the specific collection and processing of data, the reader is referred to the project report (Harmelen et al. 2003). We conclude that an abundance of data is available, but in incomparable formats and often incomplete. The data, being the basis of the analysis, could be improved in terms of accuracy and scope.

Calculation of shadow prices using cost allocation

For each EIC, the shadow price is presented in Table 4.3 for different cost allocation weights. Each shadow price is based on several measures with costs of the same order. The shadow prices for marine ecotoxicity are a factor of 100 or more lower than the other shadow prices. This is partly caused by the much higher national equivalent emission for the marine EICs, due to a high characterisation factor as a result of long residence times.

The sensitivity of the shadow prices to the cost allocation based on different weights was tested by a number of examples. The composition of the weight factors was not randomly chosen, but represents a policy

Table 4.3 Shadow prices for EICs with cost allocation according to different weight factors in € / 1,4-dichlorobenzene equivalent

Perspective (weight)	**HTP**	**FAETP**	**MAETP**	**FSETP**	**MSETP**	**TETP**
	[€ / 1,4-dichlorobenzene equivalent]					
Effect oriented (1:1:1:1:1:1)	0.042	0.107	0.00027	0.067	0.00037	1.28
Human–ecological (5:1:1:1:1:1)	0.075	0.083	0.00021	0.052	0.00028	1.21
H–E marine (5:0.4:1.6:0.4:1.6:1)	0.069	0.025	0.00026	0.016	0.00035	1.18
Compartmental (4:1:1:1:1:4)	0.065	0.059	0.00018	0.037	0.00020	1.34
Human dominant (10:1:1:1:1:1)	0.083	0.065	0.00016	0.041	0.00022	1.12
CML panel [a] (16:3:4:3:4:5) [b]	0.071	0.064	0.00022	0.040	0.00029	1.55
Human-compartmental (16:1:1:1:1:4)	**0.084**	**0.040**	**0.00010**	**0.025**	**0.00014**	**1.28**

[a] See Huppes et al. (2002/2003).
[b] Aquatic and sediment toxicity are not distinguished and each receive half of the weight factor.

perspective. For instance, in the human–ecological perspective, human toxicity (HTP: weight 5) and ecotoxicity (other EICs: each weight 1) receive equal priority. Please note that the different cost allocation does not change the total costs of measures, but only the distribution of costs over the different EICs, resulting in different prices.

The results in Table 4.3 show that the shadow price is not very sensitive to different weights, varying by a factor a 1.5 (TETP) to 2 (HTP) or 3 (other EICs), whereas the variation between the EICs involves a factor of 10,000. Of all perspectives, the compartmental perspective with a dominant human perspective is special in terms of processing as well as interpretation.

According to the theory of revealed collective preferences, the calculated shadow prices are in themselves an assessment of the present policy perspective. Hence, if this assessment is taken as a basis for cost allocation, the method is internally consistent. This approach has been explored using iterative calculations, in which the resulting shadow price is used for the calculation of the weights for the cost allocation to calculate new shadow prices. The weights are in fact shadow costs or accepted damage, being obtained by multiplying the present national equivalent emissions by the shadow prices.

Of the selected perspectives, the human–compartmental approach gives the most consistent results, in which the proportion of damage by present emissions in EICs is similar to the weights that are being used in the cost allocation. Hence, we selected the prices according to this perspective as national shadow prices for toxicity.

Discussion of environmental expenditures

The total reduction of equivalent emissions can be valued as 'shadow reduction costs' by multiplying by the shadow prices. These shadow reduction costs can be compared with the actual expenditure for an EIC as published in the annual national environmental report.(RIVM 2000) For instance, the shadow reduction costs for acidification are € 1.3 billion, using the shadow price calculated by CE (2002), compared to actual expenditures of € 0.8 billion in 2000. The actual expenditures are expected to be lower, since all technological options applied are cheaper than the shadow price. The calculation is illustrated in Table 4.4.

Table 4.4 Overview, per EIC, of emissions, reductions and shadow prices, expressed in equivalent units, sources of shadow prices, shadow costs of these reductions and actual expenditures according to the Dutch annual environmental report Milieubalans (RIVM 2000)

Environmental Impact Category	Unit	Emission 1999–2000	Net reduction 1990–2000	Shadow price	Source	Shadow reduction costs	Env. Expenditures (RIVM 2000)
		[eq]	[eq]	[€/eq]		[billion €]	[billion €]
Climate change	kt CO_2 eq	230,000	20,000[a]	€ 0.05	CE	€ 1.0[a]	€ 0.4
Acidification	kt SO_2 eq	705	333	€ 4.00	CE	€ 1.3	€ 0.8
Eutrophication	kt PO_4 eq	57	20	€ 9.00	CE	€ 0.2	€ 0.5
HTP	kt DCB eq	48,018	63,726	€ 0.084	TNO	€ 5.3	
FAETP	kt DCB eq	3,269	1,705	€ 0.040	TNO	€ 0.1	
MAETP	kt DCB eq	1,286,843	3,546,718	€ 0.0001	TNO	€ 0.4	
FSETP	kt DCB eq	5,232	4,467	€ 0.025	TNO	€ 0.1	
MSETP	kt DCB eq	956,092	3,054,486	€0.00014	TNO	€ 0.4	
TETP	kt DCB eq	689	196	€ 1.28	TNO	€ 0.2	
Total Toxicity	kt DCB eq	2,300,142	6,671,297	€ 0.0024	TNO	€ 6.5	€ 1.8
ADP	kt Sb eq	1,7		€ 0	TNO		

[a] inland measures in 2010

A brief analysis of marginal reduction cost curves for different EICs shows that the difference between reduction costs (the area under the cost curve) and the shadow reduction costs (the area under the shadow price, being the marginal costs at the reduction objective) vary per EIC. For climate change and acidification, this difference is estimated to be of the order of a factor of 2 to 3. This factor can only partly be deduced from Table 4.4, however.

The shadow reduction costs for the total of toxicity EICs are approximately € 6.5 billion. This is a factor of 3.5 higher than the environmental expenditures estimated in the annual national environmental report. This leads to the conclusion that the shadow reduction costs of toxicity seems to be of the expected order of magnitude.

This is striking, since the CML characterisation factors have been developed and updated over the last decade, so the present policy is not directly based upon the most up-to-date toxicity assessments available. Nevertheless, we conclude from our shadow price assessment that present policies do not seem to greatly contradict the knowledge and application of CML toxicity characterisation factors.

The larger the difference between the actual reduction costs and the shadow reduction costs, the steeper the marginal cost curve, indicating that marginal costs of the emission reduction options last taken, rise rapidly. This implies that the potential of relatively cheap options is becoming exhausted.

The Dutch set of shadow prices

The total set of shadow prices for the Netherlands is presented in Table 4.5. since uncertainties are quite large, we present prices rounded to 1 significant number.

The table also includes the shadow costs resulting from remaining equivalent emissions for the year 2000. According to the theory, the environmental damage is the result of the remaining emissions valued using the shadow price. The total national environmental damage for all eics considered in this paper is € 20 billion (5% of gdp). More than half of the damage is caused by climate change, while toxicity is responsible for one quarter, which is dominated by human toxicity. This is consistent with, but not completely identical to, current expenditures, which show a larger share of toxicity. This is caused by the fact that the reductions for toxicity have already reached three quarters, leaving one quarter of the environmental burden. For climate change, the opposite is true: the largest part of the emissions still remain, resulting in high shadow costs.

To our knowledge, only one other estimation of a shadow price for total toxicity has been published, by NIBE (2002). Applying this price, € 0.048 per 1,4-dcb equivalent, to the remaining total equivalent emissions, which is in fact a very crude approach since the eics are different, results in a total damage of € 320 billion this is a very high figure, but it should be noted that these costs refer to sustainable emission targets and not to present policies.

Table 4.5 Overview of rounded shadow prices for environmental impact categories including the resulting damage costs for the Netherlands in the year 2000

Environmental Impact Category	Equivalent unit	Shadow price [€ / kg equivalent]	Damage [billion €]	Source
Human toxicity – HTP	1,4-DCB eq	€ 0.08	4.0	TNO
Fresh water aquatic ecotoxicity – FAETP	1,4-DCB eq	€ 0.04	0.1	TNO
Marine aquatic ecotoxicity – MAETP	1,4-DCB eq	€ 0.0001	0.1	TNO
Fresh water sediment ecotoxicity – FSETP	1,4-DCB eq	€ 0.03	0.1	TNO
Marine sediment ecotoxicity – MSETP	1,4-DCB eq	€ 0.0001	0.1	TNO
Terrestrial ecotoxicity – TETP	1,4-DCB eq	€ 1.3	0.9	TNO
Abiotic depletion – ADP	Sb eq	€ 0	0.0	TNO
Climate change – GWP 100 y.	CO_2 eq	€ 0.05	11.5	CE
Photochemical oxidation – POCP	C_2H_2 eq	€ 2	0.4	CE
Acidification – AP	SO_2 eq	€ 4	2.8	CE
Eutrophication – EP	PO_4 eq	€ 9	0.5	CE
Ozone layer depletion – ODP	CFC11 eq	€ 30	0.03	CE

4.4 Conclusions and Recommendations

On the basis of the method we have developed to assess shadow prices, and our analysis of data on policy, measures and costs in the field of human toxicity, ecotoxicity and depletion of abiotic materials, it can be concluded that:

- Toxicity is an important environmental impact category (EIC).
- 1,4-dichlorobenzene equivalents according to CML-2 are useful when kept separate for each EIC.
- The resulting set of shadow prices is now complete for the EICs of CML-2, although the perspectives of present and 2010 policies differ for different EICs.
- The set of shadow prices can be used as an environmental and economic yardstick of present policies to assess environmental profiles and evaluate environmental measures in economically consistent and quantitative terms for cost-effective decision-making by companies and policymakers.

Based on the research conducted, we recommend to:

- apply and evaluate the present set of shadow prices and investigate further the robustness, reliability and limitations of the present method and the data on current mitigation measures for toxicity, to improve the assessment method and the quality of the shadow prices;
- extend the present CML impact assessment method with a policy version that uses equivalents with a shorter time horizon (e.g. 100 years, like Global Warming Potentials) and addresses location-specific aspects and background concentrations, thus increasing the consistency with, and therefore the quality and usefulness for, present policy development;
- maintain and update the present set of shadow prices every few years to reflect the latest policies, in order to ensure a high quality set of shadow prices.

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Cases in Agriculture

5 Conservation reconsidered: a modified input-output analysis of the economic impact of China’s land conservation policy

Fan Zhang
Harvard University, Cambridge, USA

Abstract

To estimate the economic impact of China’s land conversion policy, I present a modified input–output model, applying supply constraints to the cropping sector to reflect exogenous restrictions on land availability. Strong biophysical linkages are integrated into the model to capture heterogeneities of climate, soil and terrain conditions relevant to agricultural production. Empirical study demonstrates that this long-term land-retiring programme has evident negative impacts on the rural economy. In Western China, the net present value of total social cost is USD 487 per hectare per year or a capital equivalent of land rent of USD 1,508 per hectare over 10 years, with 5% discounting.

5.1 Introduction

China's forestry and grassland sectors have been undergoing dramatic changes. The 1998 Yangtze River flooding and recurring droughts in the Yellow River basin have heightened public awareness of the severity of Western China’s ecological degradation and its potential environmental and economic consequences. In response to nation-wide concerns over trends of ecological deterioration, the central government took the initiative in 1999 to implement a national land conversion programme called

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 129–160.

'Grain for Green' (GFG), which requires the conversion of steep cropland to forest and grasslands over the period up to 2010 to rehabilitate the country's key ecosystem services.

Landowners holding agricultural land with slopes exceeding 15 degrees are required to enrol in the 'Grain for Green' programme by signing a contract indicating the amount of land to be included. Once enrolled in the programme, landowners must agree to implement a conservation plan that provides forest or grassland cover on the land for ten years. Grazing or harvesting of forage or any other commercial activity is strictly prohibited for the duration of the contract. In exchange, farmers are entitled to receive free seeds and saplings, 50% of the one-time cost of establishing the vegetative cover, annual rental payments for ten years and long-term technical assistance. So far, more than 20 provinces, 400 counties, 5700 townships, 2.7 thousand villages, 4.1 million households and 16 million farmers have become involved, and close to 10% of China's cropland has been idled under the requirements of the programme[1].

The potential benefits of this ecological programme are significant and broad-based. The conversion of steep cropland to forest or grassland can reduce soil erosion, promote improvements to off-site water quality, provide wildlife habitats and reduce atmospheric CO2 concentrations. This programme is also justified by its obvious political attractiveness and favourable tax incentives. From the farmers' point of view, since governmental rental payments for converted land generally exceed agricultural returns from this marginal land[2], these land conversion payments apparently provide greater income support to farmers than those under existing-farm subsidising programmes[3]. At the same time, policy-makers call this programme a 'no-loss' option that provides valuable economic benefits because it results in reduced agricultural output and a higher market price, narrowing the gap between the support price and the price consumers pay, and thus reducing the cost of the existing farm subsidising programmes.

[1] Chinese State Council Act No. 367

[2] Lands with slopes greater than 15 degrees (where soils are susceptible to erosion or past erosion damage) are regarded as marginal land and are targets for retiring in this land conversion programme. Hereafter, marginal land refers to the land eligible to be enrolled in the land conversion programme.

[3] In China, the Government is the sole seller of crops in the market.. Government purchases the crops from farmers, and then sells them to consumers. Government subsidises farmers by setting the purchasing price of crops higher than the market price.

However, since agricultural production is not isolated in the economy, the consequences of this land conversion programme are more complicated than expected. As a result of the economic links between agriculture and the industries supplying its input (upstream) and processing its output (downstream), the substantial changes induced by the land conversion programme will affect the entire economy by forcing cutbacks in industries linked directly or indirectly to agricultural production. Thus, this land conversion programme is not a type of 'no regrets' policy. It accomplishes a beneficial end at the cost of welfare loss in other sectors. In addition, producers in these indirectly affected sectors suffer income loss without being compensated. Considering the interrelationships among various economic sectors, there is a need for a detailed investigation of the supply and demand relationships among many interacting agents that are related by land-based commodities and resources to assess the total social cost and to achieve a fair compensation scheme.

Cost information is also important for the eco-efficiency study of resource conservation policies. While afforestation is an effective approach for ecological conservation targets, the decision to pursue a land-use change strategy should be based at least partly on the costs of land conversion relative to those of other approaches. Although this programme has now been implemented for six years, little has been done to specifically evaluate how such programmes should be designed and for which regions. The total cost of transforming marginal agricultural land into forests remains unknown.

The purpose of this paper is to present an integrated theoretical framework to estimate the overall economic impact of the land conversion policy in China. This paper adds to earlier studies in two ways. First, it calibrates the economic impact of land-use change by accounting for its indirect effects on other sectors. Most earlier studies of the costs of land-use change have been limited to the direct costs incurred, measured by estimated expenditures on tree planting and other forestry practices, and the cost of foregoing opportunities to continue alternative uses of the land. They have thus ignored the multiplying process in the economy.

I use an input–output analysis to estimate the general equilibrium impact of structural changes to the economy. Compared to computable general equilibrium (CGE) models, input–output analysis can address the problem at lower computational cost. For a given accounting period, the basic input–output relations are represented by fixed coefficients, indicating that the physical structure of the economy in the accounting period does not automatically adapt to the structural change. This assumption is less problematic when applied to the transitional economy in China, where many industries are still state-owned and run on a highly centralised basis. By contrast, the classical CGE framework uses a selection and combination of inputs that are endogenous. The optimal solution represents input changes in response to exogenous structural change. However, there is no easy way to assess whether the changes described by the resulting combination of inputs are feasible or not. In view of these considerations, analyses in this paper rely on an input–output rather than a CGE model. Model Model

Several economic impact studies of the U.S. Conservation Reserve Programme (CRP) have been conducted using input–output analysis. Mortenson et al. (1990) evaluated the impact of the CRP on North Dakota and five subregions of the state. Martin et al. (1988) evaluated the impact of the CRP on three agriculture-dependent counties in Oregon. Hyberg et al. (1991) use an input–output model to investigate the impacts of the CRP on 5 industrial sectors at national, regional and local levels. Broomhall and Johnson (1991) estimated the regional impacts of CRP in the Southeast of the U.S.

The above standard input–output analyses assumed that all production activities are demand-driven, implying excess capacity throughout the economy; that supply is perfectly elastic in all sectors; and that an increase in demand is sufficient to stimulate increases in output and incomes. When dealing with land use, however, it will be clear that agricultural sectors do not automatically expand or shrink land requirements in direct proportion to output changes, because of limited land availability. As a result, the model derived from standard assumptions on supply response will provide multiplier estimates that are unrealistically large. In this paper, the standard input–output model is therefore modified by incorporating supply constraints on crop production activities to permit a more realistic evaluation of multiplying effects.

The second new aspect of the study is that I take the great biophysical and socio-economic heterogeneity of land use in China into account by implementing a number of fairly large and detailed geographic information databases and statistics from county level surveys. Opportunity costs of land retirement depend not only on the price and acreages of converted land, but also on the productivity, including the physical attributes of the land. Due to China's highly diverse geographical circumstances, a site-specific analysis should be carried out instead of the standard practice of taking a single-point estimate as the average amount from which to derive the overall agricultural productivity. In addition, I also consider important agronomic features of China, such as the land's multiple cropping potential, etc.

Stavins (1999) has used an econometric approach to measure the region-specific marginal cost of carbon sequestration by afforestation. The observed individual decisions regarding the use of lands for forestry or agricultural production depend on the returns offered by alternative uses, as well as the frequency of flooding, drainage and soil conditions, the natural lay of the land and the type of soil, etc. Aggregating first-order conditions for individual landowners to the county level yields relationships between county-level variables and the distribution of parcel-specific feasibility of agricultural production. However, this revealed preference approach could not be applied in this study, because landowners are required to enrol the eligible land in the programme. Since land conversion decisions are not based on market prices, land suitability or farmers' individual preferences, first-order conditions do not always hold. In this paper, I use detailed land-use databases and the agro-ecological zoning methodology, using a land resources inventory to assess the potential suitability and productivity of a particular land area for agricultural uses, depending on its soil, terrain and climate conditions, given input and management levels.

The rest of the paper is organised as follows. Section 5.2 discusses the analytical framework, assumptions and data used. Section 5.3 presents the empirical application in Gansu province and its results. Concluding comments are offered in section 5.4.

5.2 Analytical model

Economic impact analysis: concept and principles

Economic impact analysis of policy changes measures the changes in economic activity occurring in the marketplace. It can be divided into: direct economic impact (changes in the revenues of target sectors); indirect economic impact (the foregone expenditures on other sectors which are forward- or backward-linked to the targeted sectors); induced economic impacts (changes in the consumption of goods and services which are induced by changes in the income of economic agents affected by direct and indirect economic impacts).

Since the present study is based on a societal accounting stance, changes in transactions among various sectors within the economy are ignored. The principles of the analyses are described in detail below.

Governmental compensation for giving up part of the agricultural production is a fiscal transaction from the rest of the taxpayers to the farmers under a single-price system. It is not a measure of welfare loss or gain from the societal point of view. The assessment of direct economic impact should exclude the change in farmers' incomes, since it reflects the income redistribution.

1. Other economic sectors (i.e. agricultural input and processing sectors) are not compensated for their reduced level of economic activity. The revenue loss of these sectors should be considered as social welfare change.

2. The increased incomes of farmers are expected to encourage increased household consumption. On the other hand, there is an income loss in uncompensated sectors. These income changes simultaneously feed back into the household expenditure sector and tend to offset each other. The income effect on consumption also partly reflects the process of income redistribution. The measurement of change in regional economy does not include the changes induced in household consumption.

Based on these principles, I consider the following direct and indirect economic impacts.

(a) Reduced crop production. This is the most identifiable direct economic impact of the land conversion programme, deriving from the opportunity costs of tying up large acreages of agricultural land for extensive periods.

(b) Establishment and maintenance cost of the land conversion. These are the one–time establishment costs of planting trees and creating grasslands, and the annual maintenance costs.

(c) Indirect effects on other sectors. These can be traced from the reduction in crop production through the reductions in the relevant agricultural input and processing sectors to the goods and services sectors providing support to these related sectors. Agricultural inputs include farm machinery, chemical and fertiliser inputs, etc. Agricultural processing includes the primary handlers of grains and livestock, and all of the secondary handlers and manufacturers of high-value products.

A modified input–output model with supply constraints

An input–output model is a general equilibrium approach based on an accounting system of intersectoral purchases and sales. In an input–output model, connections between different sectors are described by a series of technical coefficients, which link the output of an industry to the required inputs from all the other industries in the economy. These coefficients are used to develop a system of linear equations, each of which gives the output, x_i, of a given sector i as the sum of the sector's sales to all other sectors and to the final demand y_i. For the sake of generalisability, the analysis assumes an open economy with I intermediate sectors, Q final sectors (final demand categories, including exports), and P primary sectors (value added categories and imports). The following two accounting identities hold, respectively, for total industrial outputs and inputs (matrix notation in parentheses):

$$x_i = \sum_{j=1}^{I} z_{ij} + \sum_{q=1}^{Q} y_{iq} \qquad (X = Z_i + Y_i) \qquad (5.1)$$

$$x_j = \sum_{i=1}^{I} z_{ij} + \sum_{p=1}^{P} v_{pj} \qquad (X' = i'Z + i'V) \qquad (5.2)$$

where X is an I-dimensional column vector of total output (input) of each industry; Z is an $I \times I$ matrix of intermediate inputs (outputs), Y is an $I \times Q$ matrix of the final demand on each sector; V is a $P \times I$ matrix of primary inputs and i is an identity vector, i.e. a summation vector.

The input–output model starts with output identity (5.1) and adds to it the well-known assumption of fixed input coefficients[4]:

$$z_{ij} = a_{ij} x_j \qquad (Z = A\hat{X}) \qquad (5.3)$$

[4] The assumption of fixed output coefficients is base don the assumption of cost-minimising behaviour of firms operating with a Walras-Leontief production function: $x_j = \text{Min}\,(z_{ij}/a_{ij}$ for all i; v_{pj}/c_{pj} for all $p)$.

$$v_{pj} = c_{pj} x_j \qquad (V = C\hat{X}) \tag{5.4}$$

where a_{ij} and c_{pj} are the input coefficients that measure the inputs of per unit output of sector j from intermediate sector i and primary sector p. $\hat{X}$ is an $I \times I$ diagonal matrix of total output from each industry. The sum of all input coefficients equals one, i.e. $i'A + i'C = i'$

The solution to the traditional input–output model is:

$$X = (I - A)^{-1} Y \quad \text{or} \quad \Delta X = (I - A)^{-1} \Delta Y \tag{5.5}$$

where I is the unit matrix. $(I - A)^{-1}$ is the matrix of multipliers (Leontief coefficients). Equations (5.5) allow the calculation of the total output ΔX and the change in total output, which are functions of the final demand ΔY or the change in final demand ΔY.

The above traditional input–output model assumes no resource constraints and excess capacity throughout the economy. In reality, however, production activity in the cropping sector is unable to adjust immediately to changes in other sectors, because of limited land availability. Therefore, the traditional input–output model needs to be modified to incorporate these exogenous supply constraints on agricultural production activities. After supply constraint has been applied to the cropping sector, the input and output relationships among different sectors are shown in Equation (5.6). The proof of this equation is given in Appendix A5.I.

$$\begin{bmatrix} X_{no} \\ Y_{co} \end{bmatrix} = \begin{bmatrix} P & 0 \\ R & -1 \end{bmatrix}^{-1} \begin{bmatrix} I & Q \\ 0 & S \end{bmatrix} \begin{bmatrix} \overline{Y}_{no} \\ \overline{X}_{co} \end{bmatrix} \tag{5.6}$$

where the sub-matrices are:

P is a $(n-1) \times (n-1)$ matrix composed of the first n-1 rows and the first n-1 columns of $(I$-$A)$; P represents the average expenditure propensities of sectors that are not supply-constrained.

R is a $(n-1) \times 1$ matrix composed of the last n-1 rows and the first n-1 columns of $(I$-$A)$; R represents the average expenditure propensities of unconstrained sectors on the output of the supply-constrained sector.

X_{no} is an $(n\text{-}1)$-element column vector with elements x_1 through x_{n-1}, representing the endogenous total output of sectors that are not supply-constrained.

Y_{co} is composed of element Y_n, representing the endogenous final demand of the supply-constrained cropping sector.

Q is an $(n-1)\times 1$ matrix composed of the last column and the first k rows of the matrix $-(I\text{-}A)$; the matrix Q represents the expenditure propensities of supply-constrained sectors on the output sectors that are not supply-constrained.

S is the element of $-(1-a_{nn})$; S represents the average expenditure propensities of the supply-constrained sector.

$\overline{Y}_{no}$ is an $(n\text{-}1)$-dimensional column vector composed of elements y_1 through y_k, representing the exogenous final demand on sectors that are not supply-constrained.

$\overline{X}_{co}$ is the element x_n, representing the exogenous total output of the supply-constrained cropping sector.

In the modified model, changes in exogenous final demand on the unconstrained sectors or changes in exogenous supply by the constrained sectors are met by changes in the output of the unconstrained sectors and by changes in the imports and exports of the constrained sectors.

The derived net exports T of the supply-constrained sector are given by the difference between the exogenous final demand and the endogenous final delivery:

$$T = \overline{Y}_{co} - Y_{co} \tag{5.7}$$

The reduced outputs of sectors 1 to n-1 are given by Equation (5.8).

$$\Delta x_j = x_j - x_j^0 \qquad 1 \le j \le n-1 \tag{5.8}$$

where x_j^0 is the output of sector j before the land conversion programme, while x_j is sector j's output after the programme.

Figure 5.1 is a schematic representation of the above approach, applied to a single region.

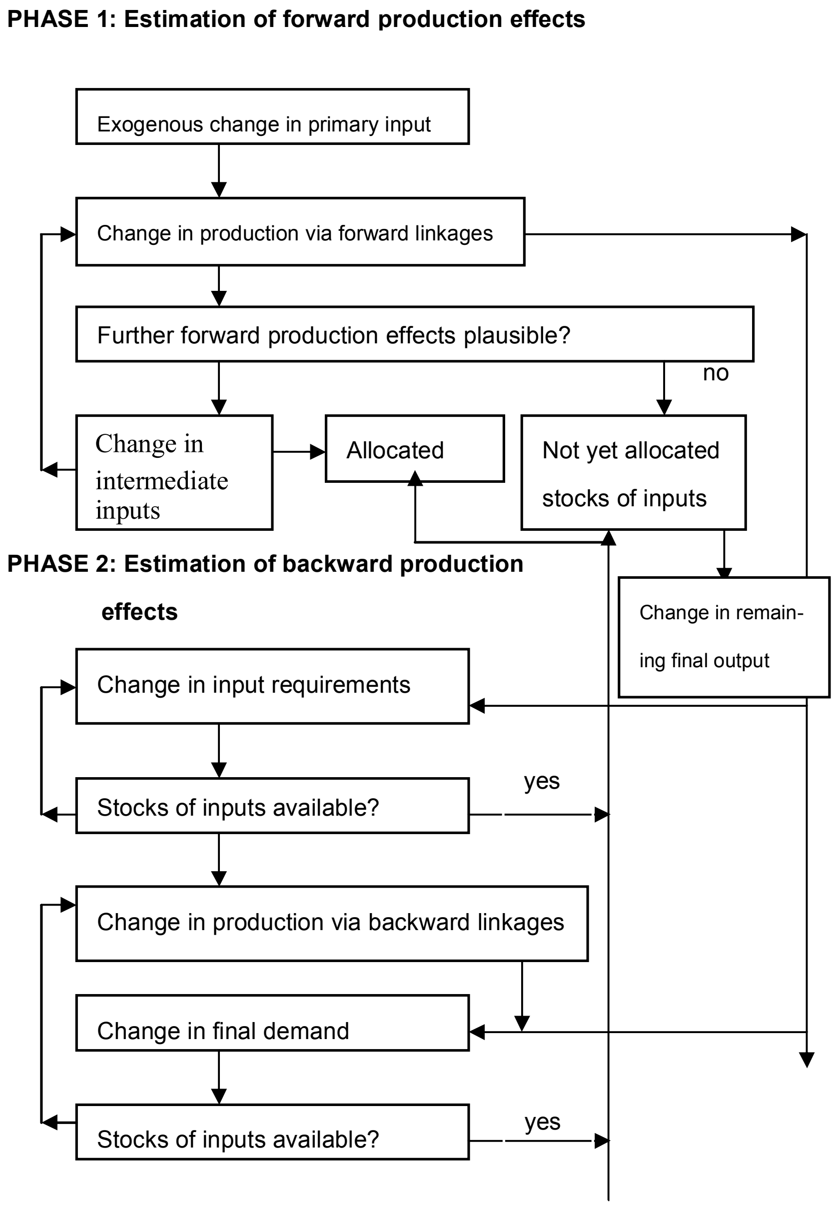

Figure 5.1 Input–output model with supply constraint for regional economic impact studies

Direct impact analysis

To estimate the direct impact of the land conversion programme on the cropping sector, I consider the amount of land to be converted and its productivity. The strong data support from a remote sensing database has allowed a detailed and spatially explicit estimation.

The survey data of the global digital elevation model (DEM) GTOPO30 allow me to determine the share of cultivated land in each county, and distinguish its terrain slopes using the seven classes described in Table 5.1[5]. Cultivated land rated as classes 6 and 7 in each grid cell of GTOPO30 represents the retiring target under the land programme. In this way, I can identify the distribution and acreage of converted land.

Table 5.1 Slope ratings of the seven slope gradient classes

Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7
0–2% (0–1^{o})	2-5% (1-3^{o})	5-8% (3-5^{o})	8-16% (5-10^{o})	16-30% (10-15^{o})	30-45% (15-25^{o})	>45% >25^{o}
Flat	Gently sloping	Undulating	Rolling	Hilly	Steep	Very steep

Note: The relationship between the percentage unit and degree unit of land slope is arctan-1 (percentage value) = value of degree. For example, arctan-1 (45%) = 25^{o}

The maximum resolution of the satellite remote sensing image is 1 km 1 km. A1-km grid cell is classified as cultivated land as long as its dominant land cover type is cultivation. Estimates based on geographical information systems (GIS) tend to overestimate the distribution of cultivated land

[5] The GTOPO30 database was established by EROS Data Centre in 1996. It provides digital elevation data in a regular grid spacing of 30 arc-seconds (approximately 1 km). GTOPO30 is derived from several raster and vector sources of topographic information. Detailed information on the characteristics of GTOPO30, including its data distribution format, data production methods and accuracy, can be found at http://edcdaac.usgs.gov/gtopo30/gtopo30.html. The terrain-slope database derived from GTOPO30 was established at IIASA. Based on neighbourhood relationships among grid cells in the GTOPO30 database, terrain slopes are calculated per 5-minute grid cell of the Digital Soil Map of the World (DSMW). Details of the algorithm used to calculate the slope distributions of cultivated land can be found in Fischer et al. (2002).

if the land is not exclusively used for agricultural purposes. To adjust for the measurement error, I cross-check the GIS data with survey data collected by the International Institute for Applied Systems Analysis (IIASA). The arithmetic to adjust for the measurement error is described in Appendix II.

Next, I assess land productivity, considering the great variety of landscapes in different locations in China. Agricultural outputs are influenced by local climate, soil and topographic characteristics of the land. They are also related to the production modes, i.e. whether the land is rain-fed or irrigated. For example, slope is one of the most important land characteristics that directly influence cropping performance. Cropping activities on sloping land are disadvantaged by the loss of applied fertiliser and fertile topsoil. Rain-fed sloping cropland suffers the most from topsoil erosion, because of its particular management style and the dynamic movement of the topsoil.

I apply an agro-ecological zones (AEZ) model to estimate land productivity under explicit recognition of the land's biophysical and socio-economic settings[6]. The application of the AEZ matching model in this study is briefly described below.

First, AEZ provides a standardised framework for characterising climate, soil and terrain conditions relevant to agricultural production. The concepts of length of growing period (LGP) and latitudinal thermal climates have been applied in zoning lands in China. Within a cropping system zone (CSZ), the climate, soil and topographic characteristics are similar. Based on the study by Zhang, China is divided into 50 cropping system zones, shown in Figure 5.2.

Second, AEZ matching procedures are used to identify crop-specific maximum output for prevailing climate, soil and terrain resources, given levels of inputs and management conditions in each CSZ land unit.

Third, assuming the same level of inputs and management, AEZ procedures compare the crop-specific output between rain-fed and irrigated

[6] The AEZ model was originally developed by IIASA and the Food and Agriculture Organisation (FAO) of the United Nations in the early 1980s and was then repeatedly used and subsequently improved in several global and national studies. Details of this model can be found in Fischer et al. (2002) or http://www.iiasa.ac.at/coolections/IIASA_Research/Research/LUC/ChinaFood/indepth/id_11.htm.

lands. Table 5.2 shows the ratios between rain-fed and irrigation yields for 14 staple food crop types for each of the 50 CSZ land units in China.

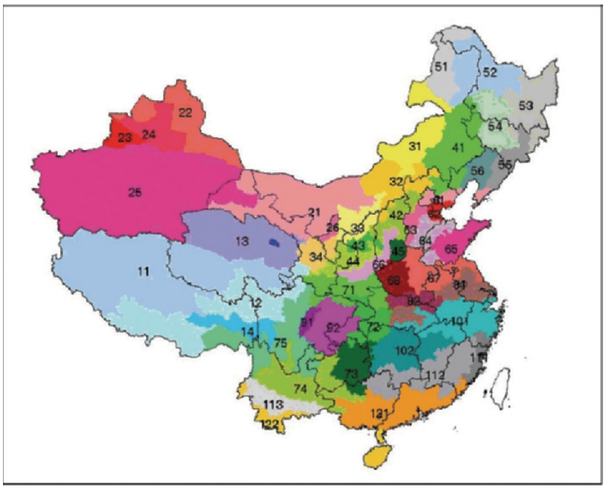

Figure 5.2 Map of China AEZ Units

Note: Each number represents one AEZ land unit with similar climate, soil, terrain resources and length of growing period.

The information on the distribution of converted land and its productivity on a county basis allows me to calculate the foregone revenue from converted land from Equation (5.9).

$$Agri-revenue=\sum_{j=1}^{J}\sum_{i=1}^{I}(yld_{ij}^{RF2}\times P_i\times GFG_{ij}\times MCI_j) \quad (5.9)$$

where i indexes the type of crop and j indexes the county; GFG_{ij} is the acreage of converted land; P_i is the market price of that crop. One would expect an increase in crop prices as a direct result of this land conversion programme. However, as I am interested in opportunity cost that would not have occurred otherwise, I use pre-programme prices to calculate foregone agricultural revenue. The price data were obtained from the China Statistics Yellow Book, 1997.

Table 5.2 Productivity relationship between rain-fed and irrigated croplands in China (14 staple food crops)

CSZ	RICE	WHEA	ROOT	MAIZ	SORG	MLLT	STCH	SOYB	COTT	OSED	CANE	BEET	VEGE	CERE
11	0.90	0.70	0.85	0.70	0.90	0.90	0.70	0.90	0.85	0.90		0.90	0.80	0.80
12	0.95	0.73	0.95	0.95	0.95	0.95	0.69	0.94	0.88	0.88		0.95	0.80	0.83
13	0.90	0.67	0.82	0.70	0.93	0.90	0.75	0.79	0.85	0.70		0.68	0.70	0.68
14	0.90	0.93	0.95	0.70	0.90	0.90	0.93	0.95	0.85	0.95		0.95	0.80	0.93
21	0.10	0.29	0.30	0.25	0.25	0.26	0.24	0.37	0.10	0.26		0.25	0.26	0.22
22	0.10	0.35	0.35	0.10	0.10	0.10	0.26	0.10	0.10	0.33		0.37	0.32	0.22
23	0.10	0.28	0.25	0.10	0.10	0.10	0.22	0.10	0.10	0.36		0.25	0.35	0.18
24	0.10	0.31	0.24	0.10	0.10	0.10	0.16	0.10	0.10	0.10		0.23	0.10	0.14
25	0.10	0.18	0.10	0.10	0.10	0.10	0.14	0.10	0.10	0.10		0.10	0.10	0.12
26	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10		0.10	0.10	0.10
31	0.10	0.38	0.37	0.10	0.10	0.10	0.39	0.10	0.10	0.10		0.10	0.10	0.38
32	0.10	0.40	0.51	0.74	0.70	0.59	0.41	0.53	0.10	0.41		0.55	0.40	0.39
33	0.10	0.35	0.32	0.10	0.10	0.31	0.29	0.10	0.10	0.32		0.31	0.32	0.24
34	0.10	0.48	0.63	0.55	0.55	0.49	0.47	0.69	0.10	0.54		0.54	0.54	0.45
41	0.27	0.40	0.43	0.41	0.55	0.46	0.41	0.40	0.10	0.39		0.37	0.37	0.34
42	0.10	0.39	0.53	0.40	0.64	0.40	0.37	0.52	0.10	0.33		0.38	0.33	0.31
43	0.10	0.46	0.60	0.43	0.45	0.48	0.43	0.63	0.10	0.40		0.42	0.40	0.39
44	0.10	0.57	0.77	0.78	0.70	0.68	0.58	0.77	0.72	0.57		0.71	0.57	0.59
45	0.40	0.49	0.81	0.49	0.67	0.63	0.58	0.65	0.10	0.49		0.52	0.49	0.52
51	0.75	0.83	0.85	0.90	0.90	0.77	0.83	0.95	0.75	0.69		0.10	0.69	0.83
52	0.72	0.75	0.91	0.91	0.90	0.75	0.73	0.95	0.75	0.74		0.74	0.75	0.77
53	0.85	0.95	0.95	0.95	0.95	0.92	0.95	0.95	0.75	0.95		0.95	0.95	0.95
54	0.49	0.62	0.76	0.74	0.95	0.67	0.62	0.79	0.75	0.67		0.71	0.66	0.62
55	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.75	0.95		0.95	0.95	0.95
56	0.35	0.60	0.76	0.72	0.72	0.77	0.72	0.71	0.92	0.69		0.62	0.65	0.70
61	0.35	0.45	0.65	0.58	0.75	0.83	0.68	0.72	0.10	0.51		0.51	0.47	0.61
62	0.22	0.28	0.42	0.52	0.81	0.81	0.67	0.67	0.10	0.43		0.23	0.31	0.59
63	0.22	0.18	0.63	0.51	0.76	0.79	0.64	0.67	0.10	0.40		0.61	0.26	0.55
64	0.37	0.41	0.64	0.55	0.78	0.79	0.65	0.66	0.36	0.48		0.66	0.27	0.58
65	0.46	0.52	0.75	0.75	0.90	0.91	0.67	0.86	0.88	0.60		0.75	0.48	0.74

Table 5.2 (cont.)

Table 5.2 (cont.)

66	0.59	0.42	0.61	0.53	0.65	0.67	0.49	0.60	0.68	0.36		0.71	0.35	0.52
67	0.68	0.78	0.90	0.90	0.93	0.95	0.83	0.94	0.92	0.84	0.80	0.89	0.81	0.89
68	0.79	0.76	0.91	0.88	0.88	0.95	0.79	0.93	0.92	0.81	0.80	0.89	0.80	0.87
71	0.95	0.95	0.95	0.95	0.85	0.95	0.95	0.95	0.95	0.95	0.90	0.95	0.95	0.95
72	0.95	0.95	0.95	0.95	0.92	0.95	0.95	0.95	0.94	0.95	0.90	0.95	0.95	0.95
73	0.95	0.95	0.95	0.95	0.88	0.95	0.95	0.95	0.93	0.95	0.90	0.95	0.95	0.95
74	0.95	0.90	0.94	0.95	0.92	0.95	0.90	0.95	0.95	0.94	0.95	0.95	0.94	0.95
75	0.95	0.94	0.95	0.95	0.95	0.95	0.94	0.95	0.89	0.95	0.95	0.95	0.95	0.95
81	0.95	0.95	0.95	0.95	0.88	0.95	0.95	0.95	0.95	0.95	0.90	0.95	0.95	0.95
82	0.95	0.95	0.95	0.95	0.86	0.95	0.95	0.95	0.95	0.95	0.90	0.95	0.95	0.95
91	0.82	0.86	0.95	0.95	0.81	0.95	0.86	0.95	0.95	0.95	0.90	0.95	0.89	0.94
92	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.90	0.95	0.95	0.95
101	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.90	0.95	0.95	0.95
102	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.90	0.95	0.95	0.95
111	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95
112	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95
113	0.85	0.86	0.89	0.94	0.92	0.95	0.89	0.95	0.94	0.92	0.72	0.92	0.91	0.94
121	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95	0.95
122	0.92	0.93	0.95	0.95	0.93	0.95	0.95	0.95	0.95	0.80	0.95	0.91	0.92	0.95

MCI is the multi-cropping index. 'Multi-cropping' means utilising land for more than one cropping enterprise at the same time. With limited land resources, high levels of multi-cropping are quite common in China. *MCI* measures the total production potential.

One way to assess *MCI* is to refer to historical survey data for converted lands. Considering the tremendous data requirements in a nationwide land-use analysis, I refer instead to the agro-climatic attributes already calculated during AEZ analysis to decide the multiple cropping zones under rain-fed conditions. Multiple cropping zones are identified by matching both growing cycle and temperature requirements to the cultivation of an individual crop. Under rain-fed conditions, this period is approximated by the number of days during which both temperature and moisture conditions permit crop growth.

Total social costs

If the converted land is assumed to be placed under reserve for at least 10 years, the net present value of the total social cost of the programme is:

$$NPV = \sum_{t=0}^{9} \frac{\Delta x_{1t} + \Delta x_{2t} + \Delta x_{3t}}{(1+r)^t} \tag{5.10}$$

where Δx_{1t}, Δx_{2t} and Δx_{3t} are the reduced output value from the agriculture sector, the reduced output value from all other sectors related to agricultural activities and the establishment and maintenance costs in year *t*, respectively. In this study, the discount rate r is assumed to be 5%.

5.3 Empirical study in Gansu Province

I have applied the above analytical framework to the case of Gansu, a province located in China's western inland region. Gansu has a land area of 45.5 million hectares, including 3.53 million hectares of cultivated land, approximately 7.8% of the total territory. Its total population is 25.62 million people, of whom about 78.9% are engaged in agriculture. Gansu presents a broad range of land-use patterns and biophysical conditions. With large steep areas being cultivated, it also provides many opportunities for land conversion.

Before the land conversion programme, cultivated land in Gansu was almost exclusively used for maize and wheat production, the two major crop types considered in this study. After conversion, 95% of the land placed under reserve is covered by native grass or bioenvironmental trees with no direct market value. (Tang, et al., 2001)

Reduced return from crop production

I use digital elevation data for Gansu Province from GTOPO30 to get the cropland slope rating. Figure 5.3 depicts these elevation data in a digital map, while Figure 5.4 shows the classification of land-use types. The terrain slopes of cultivated land were obtained from the overlaps between Figures 5.3 and 5.4; these are shown in Figure 5.5.

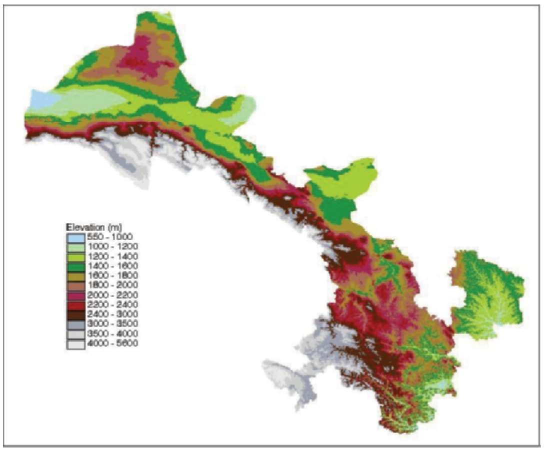

Figure 5.3 Elevation Distribution in Gansu Province
Source: GTOPO30 database (EROS Data Center, 1998)

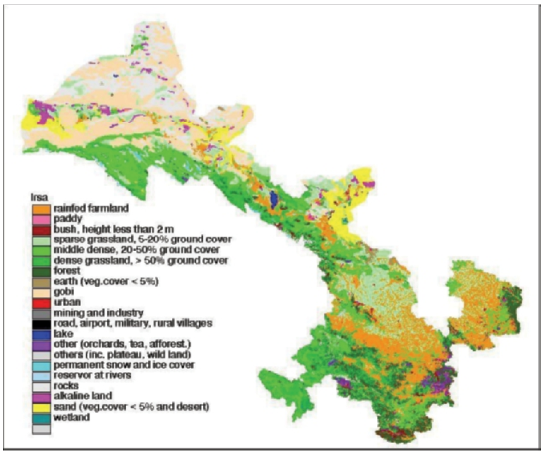

Figure 5.4 Land-use Types in Gansu Province
Source: IIASA China county database, 2000

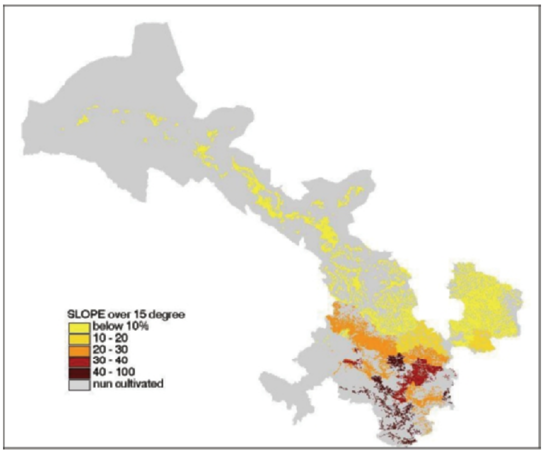

Figure 5.5 Terrain slopes and distribution of converted land

Table 5.3 Land eligible for conversion in Gansu Province 2000–2010

Counties	Cultivated area [ha]	Acreage of converted land [ha]	Percentage of converted land
Lanzhou	214,687	7,588.56	0.0353
Jiayuguan	2,843	0	0
Jinchang	46,533	173.98	0.0037
Baiyin	298,680	6,253.72	0.0209
Tianshui	383,734	110,199.91	0.2872
Jiuquan	112,026	4.57	4.0759E-05
Zhangye	187,233	910.32	0.0049
Weiwu	259,667	3,944.83	0.0152
Dingxi	518,213	89,734.28	0.1732
Longnan	290,540	109,966.49	0.3785
Pingliang	395,466	42,545.00	0.1076
Qingyang	437,568	19,177.75	0.0438
Linxia	145,094	35,884.47	0.2473
Gannan	68,210	22,844.71	0.3349
Gansu	**3,360,494**	**449,228.54**	**0.1337**

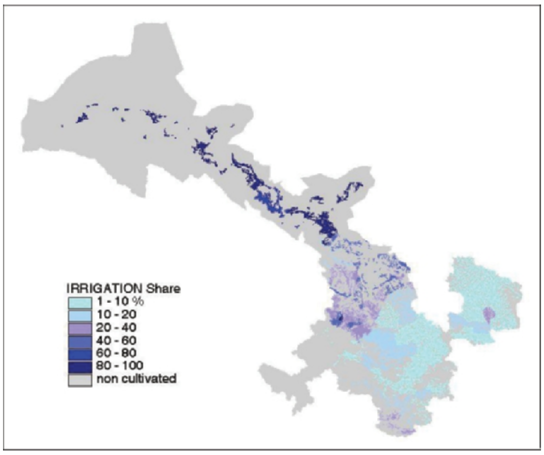

Figure 5.6 Distribution of irrigated cropland in Gansu Province
Source: IIASA China County Database, 2000

Table 5.3 presents the acreage of convertible land for each county in Gansu province, adjusted for measurement error. Specifically, it shows that a total of 13.37% of the arable land in Gansu will be taken out of production under the land conversion policy.

Gansu province is divided into 9 cropping system zones, as shown in Figure 5.2. For each zone, there is an index specifying the crop-specific output ratio between rain-fed and irrigated land. These indexes are highlighted in yellow in Table 5.2. Data on the distribution of irrigation and rain-fed land have been compiled from IIASA China County Database 2000, and are shown in Figure 5.6.

Next I calculate the *MCI* index. The *MCI* of converted land is likely to be lower than the average value in that region, because the land that is eligible for conversion is generally of low quality and has lower production potential than less steep rain-fed land. As a simplified assumption, I take the *MCI* of converted land to be equal to 0.95 of the average *MCI* of the same region.

Table 5.4 shows agricultural revenues foregone under the land conversion programme.

Table 5.4 Agricultural revenue lost from the converted land in Gansu Province

Counties	Average yields in one county [ton/ha]		Revenue
	Per ha of maize [tons]	Per ha of wheat [tons]	Total [USD$]
Lanzhou	3.056	0.918	1,871,686
Jiayuguan	2.048	1.563	0
Jinchang	1.416	1.196	34,212
Baiyin	2.879	1.0124	1,294,481
Tianshui	2.079	0.742	23,853,873
Jiuquan	1.732	1.517	1,273
Zhangye	2.146	1.481	192,748
Weiwu	0.742	1.159	612,746
Dingxi	1.542	0.842	15,868,224
Longnan	3.061	1.343	37,074,180
Pingliang	2.182	0.848	10,518,117
Qingyang	2.611	0.555	3,867,851
Linxia	2.621	1.348	10,451,947
Gannan	0.956	1.183	5,829,168
Gansu (Average)	**2.135**	**1.096**	**111,470,507**

Note: Estimation based on 1997 prices

Indirect economic impacts on other sectors

The estimation of the indirect economic impacts of the land conversion programme is based on the latest available Gansu input–output table. The original table includes 124 sectors, 5 of which are in agriculture, 84 in industry, 1 in construction, 9 in transport and communication, and 25 in service sectors. To capture the fundamental distribution impacts of the land conversion programme while simplifying the calculation, I have aggregated the original table into 14 sectors.

The agricultural sectors have been maintained in full detail, including five sub-sectors: cropping, forestry, livestock, fishery and others. The 'value-added' categories include the following: capital depreciation, labour compensation, taxes and profits. 'Final use' comprises six categories: peasant, non-peasant and government consumption, fixed investment, inventory changes and net exports. The negative numbers in the import column reflect a negative trade balance. The error column is included to balance the table. For the complete input–output table and the technical coefficients between sectors, see Tables 5.5 and 5.6.

Estimations of the indirect impacts of the land conversion programme are presented in Table 5.7. This land–retiring programme would generally produce negative effects on various businesses and economic sectors. The direct effect of a USD 111.47 million loss of revenue in the cropping sector would result in about USD 98.42 million of revenue loss in all the other sectors in the province – an overall multiplier of 0.88, which amounts to 0.7% of the GNP[7].

Establishment and maintenance cost

It has been estimated that the cost of establishment and maintenance of the vegetative cover in Gansu Province averaged USD 49.38 per hectare in the first year and USD 4.94 per hectare in the following years (GPCCPIP, 2003). The total fixed cost of the implementation of conservation practices is therefore USD 8.98 million per year.

Table 5.8-9 summarises the three aspects of economic impacts estimated above. Applying a discount rate of 5% and a 10-year time horizon, the present value of the capital equivalent per hectare associated with the land conversion programme is USD 9,313. Taking 0.4 as the output elasticity of land (Albersen et al. 2002), the annual 'rent' for land comes to USD 487 per hectare and the corresponding total land capital value amounts to USD 1,508 per hectare.

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[7] The multiplier is defined as the total gross output change in other sectors divided by the change in the cropping sector.

Table 5.5 Input–output table for Gansu, 1997 (in thousand Yuan)

	Cropping	Forestry	Livestock	Fishery	Other agr	Energy
Cropping	2560897.80	3198.50	1859854.80	1729.00	134824.50	1089.60
Forestry	4157.40	118557.30	1675.90	71.80	5010.20	907.40
Livestock	0.00	0.00	106776.30	0.00	0.00	0.00
Fishery	0.00	0.00	0.00	1589.00	0.00	0.00
Other agr	37505.30	7072.00	104519.90	8818.40	456335.30	43.80
Energy	1081619.70	22556.20	54674.90	1203.80	74312.10	12112569.50
Food processing	253286.30	2445.90	586360.90	637.60	24994.10	19259.50
Fertiliser	2562086.90	54140.50	0.00	148.50	0.00	11.30
Industry	1169040.90	52339.60	126195.50	3653.40	285297.80	2739397.30
Construction	0.00	0.00	0.00	0.00	0.00	93866.80
TransportGW	795776.70	17638.30	120717.90	517.40	65920.10	937459.80
Trade WR	1222923.20	45341.30	385650.20	2619.10	131716.60	1807139.60
Restaurant	0.00	0.00	0.00	0.00	0.00	24076.80
Services	742963.20	29794.80	147885.00	433.50	46526.10	969227.90
Intermediate Demand	10430257.40	353084.40	3494311.30	21421.50	1224936.80	18705049.30
Capital	1192481.50	43622.20	711214.60	9026.00	61477.70	2206741.40
Labour	8112387.00	285042.00	4410732.70	36723.00	418219.60	3515986.70
Net tax	1072628.50	33173.20	532004.20	7291.00	55297.70	2735086.00
Surplus	-93443.40	27141.20	214382.20	18626.50	-166750.80	4294493.50
SUM	**20714311.00**	**742063.00**	**9362645.00**	**93088.00**	**1593181.00**	**31457356.90**

Table 5.5 (cont.)

5.4 Conclusion

A general equilibrium analysis is therefore necessary to understand the comprehensive socioeconomic consequences of the land conversion programme and to achieve the same policy objectives more efficiently, equitably and sustainably. For example, based on the general equilibrium analysis, the government may consider introducing an ecological tax to divide the burden of creating ecological reserves between the upstream economy and the downstream taxpayers. The government should also encourage the local communities to proactively create new economic links by attracting new firms to the region to offset losses in the existing sectors. For example, since reserved land provides better habitats for fish and

Table 5.5 (cont.)

Cropping	2488382.90	34.70	687218.00	0.00	0.00	2497.50	223118.70
Forestry	0.00	0.00	27525.60	0.00	0.00	0.00	39122.80
Livestock	716281.00	0.00	432036.90	0.00	0.00	4513.10	149284.00
Fishery	0.00	0.00	0.00	0.00	0.00	10885.90	275964.50
Other agr	680.40	8.40	27977.60	0.00	0.00	0.00	3769.10
Energy	606541.20	880876.90	8481065.80	457607.50	1874693.30	341723.30	231968.20
Food processing	729787.60	26.30	23155.70	0.00	7709.70	88343.30	2555073.40
Fertiliser	1117.80	7681.30	55876.00	0.00	199.60	17960.50	2558.60
Industry	1114522.70	396369.20	22057490.60	8828827.60	967675.80	806038.70	239557.90
Construction	6477.80	1685.80	51189.40	294946.10	262183.90	132931.90	0.00
TransportGW	349405.10	74591.90	2657224.80	925365.10	735868.80	107591.10	59950.30
Trade WR	751651.80	165231.00	4219529.30	982601.80	425009.90	177775.10	462373.10
Restaurant	27927.00	2068.70	124526.70	40510.20	67142.10	245603.30	0.00
Services	429904.70	86367.60	2904317.50	230234.00	618900.10	2515393.80	167024.70
Intermediate Demand	7222680.00	1614941.80	41749133.90	11760092.30	4959383.20	4451257.50	4409765.30
Capital	490397.10	151532.10	**2777245.20**	**319005.10**	**870045.90**	**1387742.80**	**3812.40**
Labour	1346535.60	258461.30	**10542197.00**	**2872996.30**	**2143966.00**	**3722022.80**	**1188764.50**
Net tax	976194.00	58182.70	**4295254.40**	**647002.60**	**185520.60**	**410806.30**	**378748.50**
Surplus	569589.80	136707.10	**6314799.20**	**-711099.30**	**1455450.10**	**6080853.80**	**-124819.50**
SUM	**10605396.50**	**2219825.00**	**65678629.70**	**14887997.00**	**9614365.80**	**16052683.20**	**5856271.20**

wildlife, communities may attempt to upgrade recreational services to the local population and attract tourists from outside the region.

The results of the present study have to be interpreted with some caution, for several reasons. One of the caveats is that the static input–output analysis does not consider possible adaptive behaviours of economic actors and the resulting price consequences under the condition of increasing scarcity of land resources.

The current input–output analysis assumes that the local economy is not able to reallocate available resources between the agricultural and non-agricultural sectors. Thus, the estimate is based on a static 'snapshot' of what is, in fact a complex, dynamic system. During the relatively long period of transition, adjustments might be made to the industrial structure,

Table 5.6 Input–output table for Gansu, 1997 (in thousand Yuan)

	Service	Intermed. Demand	Peasant	Non-peasant	Institutions	Investment
Cropping	588.90	7963434.90	8372460.10	1208910.80	215044.30	0.00
Forestry	3378.00	200406.40	32734.40	47234.00	5432.00	464570.10
Livestock	1125.10	1410016.40	2843825.70	157570.50	123926.00	637326.70
Fishery	0.00	288439.40	35637.20	125645.00	4216.80	0.00
Other agr	0.00	646730.20	397697.10	182766.00	5470.80	0.00
Energy	2390893.51	28612305.91	501224.10	389346.70	152211.60	0.00
Food processing	122701.42	4413781.72	2981832.20	4154751.40	925655.80	0.00
Fertiliser	196.70	2701977.70	0.00	0.00	0.00	0.00
Industry	6623294.21	45409701.21	3147452.60	4462204.90	515580.50	6026339.10
Construction	1115983.40	1959265.10	8427.80	41192.40	0.00	12879081.70
TransportGW	664051.70	7512079.00	350448.00	191284.20	32372.30	2158171.50
Trade WR	1446383.60	12225945.60	1711564.80	1576681.50	190016.10	452934.80
Restaurant	781767.90	1313622.70	956026.40	1039823.00	2546799.10	0.00
Services	7018041.30	15907014.20	5644300.90	4544237.50	11886734.70	466199.90
Intermediate Demand	20168405.74	130564720.44				
Capital	4255782.80	14480126.80				
Labour	11506197.30	50360231.80				
Net tax	2220130.60	13607320.30				
Surplus	-1605051.60	16410878.80				
SUM	36545464.84	225423278.14	26983631.30	18121647.90	16603460.00	23084623.80

Table 5.6 (cont.)

and new forward linkages in the economy could emerge. Changes in technology, individual preferences and the redistribution of labour supply are also possible (as labourers are freed up from agricultural production). Hence, the above model may overestimate the total social cost. The conclusion is also affected by the fact that dynamic price movements for commodities were not incorporated into the modelling framework, leading to an ambiguous effect on the final results.

Table 5.6 (cont.)

Cropping	3772839.40	1686436.30	-2504805.90	-8.90	12750885.00	20714311.00
Forestry	7700.00	19289.20	-35303.10	0.00	541656.60	742063.00
Livestock	858123.20	3439158.60	-107302.00	-0.10	7952628.70	9362645.00
Fishery	-88059.00	42434.60	-315226.00	0.00	-195351.40	93088.00
Other agr	99420.00	265434.20	-4337.30	0.00	946450.80	1593181.00
Energy	222937.90	9032160.40	-7452827.80	-1.90	2845052.90	31457356.91
Food processing	1304029.40	2100020.40	-5274674.80	0.38	6191614.40	10605396.50
Fertiliser	210913.80	217283.10	-910475.20	-0.10	-482278.30	2219699.30
Industry	8438985.90	21734486.30	-24059973.40	3978.30	20265075.90	65678755.41
Construction	0.00	0.00	0.00	30.00	12928701.90	14887997.00
TransportGW	1037236.70	1349708.70	-3016933.80	-0.80	2102287.60	9614365.80
Trade WR	166544.00	1246234.20	-1517238.10	0.30	3826737.30	16052683.20
Restaurant	0.00	0.00	0.00	0.00	4542648.50	5856271.20
Services	0.00	949331.80	-2852357.00	2.80	20638447.80	36545464.80
Intermediate Demand						
Capital						
Labour						
Net tax						
Surplus						
SUM	16030671.30	42081977.80	-48051454.40	3999.98		

Source: Gansu Statistic Bureau

The future work is to sort out, through a set of case studies, the most important dynamic effects of China's land conversion process. Efforts will be focused on developing scenarios representing different economic and social changes and calculating their effects on the overall economy, including the changes in technology used in different sectors, the relative sizes of different sectors, labour supply and commodity prices.

Table 5.7 Intermediate coefficients (A-matrix) of Gansu's economy in 1997

	Forestry	Livestock	Fishery	Other agr	Energy	Food	Fertiliser	Industry	Con-struction	Transp. GW	TradeWR	Restaurant	Service	Cropping
Forestry	0.15977	0.00018	0.00077	0.00314	0.00003	0.00000	0.00000	0.00042	0.00000	0.00000	0.00000	0.00668	0.00009	0.00020
Livestock	0.00000	0.01140	0.00000	0.00000	0.00000	0.06754	0.00000	0.00658	0.00000	0.00000	0.00028	0.02549	0.00003	0.00000
Fishery	0.00000	0.00000	0.01707	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00068	0.04712	0.00000	0.00000
Other agr	0.00953	0.01116	0.09473	0.28643	0.00000	0.00006	0.00000	0.00043	0.00000	0.00000	0.00000	0.00064	0.00000	0.00181
Energy	0.03040	0.00584	0.01293	0.04664	0.38505	0.05719	0.39682	0.12913	0.03074	0.19499	0.02129	0.03961	0.06542	0.05222
Food processing	0.00330	0.06263	0.00685	0.01569	0.00061	0.06881	0.00001	0.00035	0.00000	0.00080	0.00550	0.43630	0.00336	0.01223
Fertiliser	0.07296	0.00000	0.00160	0.00000	0.00000	0.00011	0.00346	0.00085	0.00000	0.00002	0.00112	0.00044	0.00001	0.12369
Industry	0.07053	0.01348	0.03925	0.17907	0.08708	0.10509	0.17856	0.33584	0.59302	0.10065	0.05021	0.04091	0.18123	0.05644
Construction	0.00000	0.00000	0.00000	0.00000	0.00298	0.00061	0.00076	0.00078	0.01981	0.02727	0.00828	0.00000	0.03054	0.00000
Transport GW	0.02377	0.01289	0.00556	0.04138	0.02980	0.03295	0.03360	0.04046	0.06216	0.07654	0.00670	0.01024	0.01817	0.03842
Trade WR	0.06110	0.04119	0.02814	0.08268	0.05745	0.07087	0.07443	0.06425	0.06600	0.04421	0.01107	0.07895	0.03958	0.05904
Restaurant	0.00000	0.00000	0.00000	0.00000	0.00077	0.00263	0.00093	0.00190	0.00272	0.00698	0.01530	0.00000	0.02139	0.00000
Services	0.04015	0.01580	0.00466	0.02920	0.03081	0.04054	0.03891	0.04422	0.01546	0.06437	0.15670	0.02852	0.19204	0.03587
Cropping	0.00431	0.19865	0.01857	0.08463	0.00003	0.23463	0.00002	0.01046	0.00000	0.00000	0.00016	0.03810	0.00002	0.12363

Table 5.8 Indirect economic impacts of land conversion in Gansu Province

Economic sectors	Pre-programme TGO [USD]	TGO changes after programme [USD]	TGO changes per ha [USD]
Forestry	90,495,488	-46,000	-0.102
Livestock	1,141,785,976	-300,000	-0.668
Fishery	11,352,195	-35,000	-0.078
Other agriculture	194,290,366	-307,000	-0.683
Energy	3,836,263,038	-27,456,000	-61.118
Food processing	1,293,341,037	-1,891,000	-4.209
Fertiliser	270,710,366	-13,872,000	-30.880
Industry	8,009,588,989	-22,929,000	-51.0408
Construction	1,815,609,390	-763,000	-1.698
Transport GW	1,172,483,634	-7,478,000	-16.646
Trade WR	1,957,644,293	-11,829,000	-26.332
Restaurant	714,179,415	-552,000	-1.229
Services	4,456,764,000	-10,957,000	-24.391
Total		**-98,415,000**	**-219.076**

Note: TGO: Total Gross Output

Table 5.9 Annual/total social cost of GFG in Gansu Province

	Per ha per year [$]	Grand total per year [$]
A. Reduced return on crop production	248.14	111,470,507
B. Secondary effects on other sectors	219.08	98,415,000
C. Maintenance of vegetative cover	20.00	8,984,571
A+B+C	487.21	218,870,078

Note: Estimation based on 1997 prices

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Appendix

I. Proof of supply-constrained input-output model

The traditional input-output relationship of an n-sector economy as shown in Equation (5.5) is rewritten as a set of equations in (A5.1). The items in the equation have been arranged in such a way that the outputs of the first n-1 sectors and the supply to the last sector (cropping sector) are endogenous:

$$\begin{aligned}
(1-a_{11})X_1 - a_{12}X_2 - \ldots - a_{1n}X_n &= Y_1 \\
-a_{21}X_1 + (1-a_{22})X_2 \ldots - a_{2n}X_n &= Y_2 \\
&\vdots \\
-a_{n1}X_1 - a_{n2}X_2 \ldots + (1-a_{nn})X_n &= Y_n
\end{aligned} \tag{A5.1}$$

I rewrite (A5.1) by moving endogenous variables to the right-hand side of the equations and exogenous variables to the left-hand side. I use overbar to indicate exogenous variables.

$$\begin{aligned}
(1-a_{11})X_1 - a_{12}X_2 - \ldots - a_{1n-1}X_{n-1} + 0Y_n &= \overline{Y}_1 + 0\overline{Y}_2 \ldots + a_{1n}\overline{X}_n \\
-a_{21}X_1 + (1-a_{22})X_2 - \ldots - a_{2n-1}X_{n-1} + 0Y_n &= 0\overline{Y}_1 + \overline{Y}_2 \ldots + a_{2n}\overline{X}_n \\
&\vdots \\
-a_{n1}X_1 - a_{n2}X_2 - \ldots - a_{nn-1}X_{n-1} - Y_n &= 0\overline{Y}_1 + 0\overline{Y}_2 \ldots - (1-a_{nn})\overline{X}_n
\end{aligned} \tag{A5.2}$$

Rewriting (A5.2) in matrix notation yields the following:

$$\begin{bmatrix} (1-a_{11}) & -a_{12} & \cdots & 0 \\ -a_{21} & (1-a_{22}) & \cdots & 0 \\ & \cdots & & \\ -a_{n1} & -a_{n2} & \cdots & -1 \end{bmatrix} \begin{bmatrix} X_1 \\ X_2 \\ \vdots \\ Y_n \end{bmatrix} = \begin{bmatrix} 1 & 0 & \cdots & a_{1n} \\ 0 & 1 & \cdots & a_{2n} \\ & & \cdots & \\ 0 & 0 & \cdots & -(1-a_{nn}) \end{bmatrix} \begin{bmatrix} \overline{Y}_1 \\ \overline{Y}_2 \\ \vdots \\ \overline{X}_n \end{bmatrix} \tag{A5.3}$$

Naming the square matrices M and N, respectively, and rearranging them, I get:

$$\begin{bmatrix} X_1 \\ X_2 \\ \vdots \\ Y_n \end{bmatrix} = M^{-1}N \begin{bmatrix} \overline{Y}_1 \\ \overline{Y}_2 \\ \vdots \\ \overline{X}_n \end{bmatrix} \tag{A5.4}$$

(A5.4) can be further written as

$$\begin{bmatrix} X_{no} \\ Y_{co} \end{bmatrix} = \begin{bmatrix} P & 0 \\ R & -1 \end{bmatrix}^{-1} \begin{bmatrix} I & Q \\ 0 & S \end{bmatrix} \begin{bmatrix} \bar{Y}_{no} \\ \bar{X}_{co} \end{bmatrix} \tag{A5.5}$$

Specifications of (A5.5) can be found in section II.

II. Adjusting the measurement error of remote sensing

The estimated acreage of cultivated land based on GIS data is usually larger than the actual acreage. This is because the resolution of satellite images is limited (1 km×1 km). When a 1 km grid cell is classified as cultivated land by remote sensing data, this only refers to the dominant land-use type within that grid cell. To adjust for the measurement error, I have compared the GIS data with statistics derived from surveys. The data-adjusting arithmetic can be described as following:

$$GFG\%_j = (AG6_j + AG7_j) / AG_j \tag{A5.6}$$

$$GFG_j = AS_j \times GFG\%_j \tag{A5.7}$$

where j indexes the county; AG_j and AS_j are the area of cultivated land in county j estimated by GIS and the survey data, respectively. $AG6_j$ and $AG7_j$ are the total cropland areas with slope classes 6 and 7, according to the definition in Table 5.1, as determined using GIS data. GFG_j is the adjusted acreage of total convertible land in county j.

III. Estimating the Potential Output of the Converted Land

Equations for estimating the potential output of the converted land are presented below.

$$yld_{ij} = Q_{ij} / A_{ij} \tag{A5.8}$$

$$yld_{ij} = yld_{ij}^{IR} \times A^{IR}\%_j + [yld_{ij}^{RF1} \times (1 - A^{IR}\%_j) \times (1 - A^{RF2}\%_j) + yld_{ij}^{RF2} \times (1 - A^{IR}\%_j) \times A^{RF2}\%_j)]$$

$$yld_{ij}^{RF1} / yld_{ij}^{IR} = f(CSZ_{ij}) = R1_{ij} \qquad \text{(A5.9)}$$

$$yld_{ij}^{RF2} / yld_{ij}^{RF1} = R2_{ij} = 0.67 \qquad \text{(A5.10)}$$

$$A^{RF2}\%_j = \frac{GFG_j}{AS_j \times (1 - A^{IR}\%_j)} \qquad \text{(A5.11)}$$

$$yld_{ij}^{IR} = \frac{yld_{ij}}{A^{IR}\%_j + (1 - A^{IR}\%_j)[R1_{ij} \times (1 - A^{RF2}\%_j) + R1_{ij} \times R2_{ij} \times A^{RF2}\%_j)]} \qquad \text{(A5.12)}$$

$$= \frac{yld_{ij}}{A^{IR}\%_j + (1 - A^{IR}\%_j) \times R1_{ij} \times (1 - 0.33 \times A^{RF2}\%_j)}$$

$$yld_{ij}^{RF1} = R1_{ij} \times yld_{ij}^{IR} \qquad \text{(A5.13)}$$

$$yld_{ij}^{RF2} = R2_{ij} \times yld_{ij}^{RF1} = R1_{ij} \times R2_{ij} \times yld_{ij}^{IR} \qquad \text{(A5.14)}$$

where i indexes the type of crop and j indexes the county; Q_{ij} is the total output of that crop in that county; A_{ij} is the cultivated land area; yld_{ij} is the average per hectare yield; $yld^{IR}{}_{ij}$ is the per hectare yield from irrigated cropland; $yld^{RF1}{}_{ij}$ is the per hectare yield from non-steep rain-fed cropland; $yld^{RF2}{}_{ij}$ is the per hectare yield from steep rain-fed cropland; $A^{IR}\%j$ is the share of irrigated land in the total arable land; $A^{RF2}\%j$ is the share of steep rain-fed cropland in the total rain-fed land; CSZ_{ij} is the crop-specific (crop i) AEZ land-unit based (county j) cropping system zone index; f represents the output relationship between the rain-fed and irrigation production modes, which is a function of the CSZ index; $R1_{ij}$ is the ratio of output i from rain-fed cropland to that of irrigated cropland; $R2_{ij}$ is the ratio of output i from steep rain-fed cropland to that of non-steep rain-fed cropland j. Q_{ij}, A_{ij} and $A^{IR}\%j$ are all survey data provided by Centre for Chinese Agricultural Policy of the Chinese Academy of Sciences, and the IIASA China county database.

Based on rule of thumb, Equation (A5.10) says that the annual output from steep rain-fed cropland is 0.67 times that from flat rain-fed cropland. To solve the value of yld_{ij}^{RF2}, plug Equations (A5.9) and (A5.10) into (A5.8) and there is only one unknown variable yld_{ij}^{IR} left, which can be represented by Equation (A5.12). Plugging the value of yld_{ij}^{IR} back into Equations (A5.9) and (A5.10), one can solve Equations (A5.13) and (A5.14). The solution to Equation (A5.14) is simply the potential annual output of crop *i* from converted land in county *j*.

The type of data used and the framework for the above calculation are described in Figure A5.1.

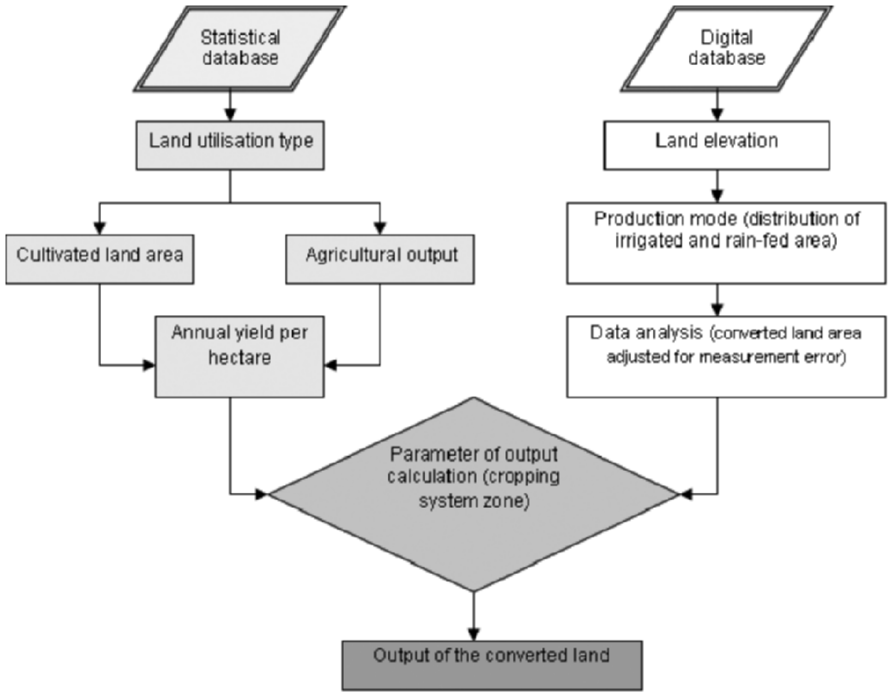

Figure A5.1 Analytical model for estimating cropping output from converted land

Cases in Industry

6 Eco-efficiency in redesigned extended supply chains; furniture as an example

Ottar Michelsen
Department of Industrial Economics and Technology Management, Norwegian University of Science and Technology (NTNU), Trondheim, Norway

Abstract

This paper shows how the eco-efficiency concept can be used to evaluate value and environmental performance when considering different scenarios for redesigning extended supply chains (ESCs). Results from a case study on furniture production in Norway are used to illustrate the concept.

An extended supply chain includes all processes necessary for production, use and end-of-life treatment of a product. The environmental performance of the products was assessed using LCA, and value performance was measured as life cycle cost. Instead of calculating absolute values using a traditional eco-efficiency ratio, relative values for different scenarios were calculated and presented graphically in an XY-diagram. This clearly visualises the alternatives that have the best environmental and value performance.

Six different scenarios were developed to assess how the performance of an existing ESC can be improved. The eco-efficiency for each scenario was compared with the present ESC. The results show that there is large and realistic potential for environmental improvements in the extended supply chain without an equivalent increase in life cycle costs.

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 163–179.

6.1 Introduction

The growing concern for the environmental dimension of business strategy is resulting in a greater focus on environmental management (e.g. Porter and van der Linde 1995; Noci and Verganti 1999; Cramer 2000; Hall 2000; Ammenberg and Hjelm 2003; Banerjee et al. 2003; Hunkeler et al. 2004). More and more companies have also realised that this has consequences not only for the activities within the company, but for the entire supply chain (e.g. Lamming and Hampson 1996; Noci and Verganti 1999; Clift and Wright 2000).

The increased focus on environmental performance in companies has a manifold origin. Pressure from customers and legislation have often been identified as the two most important drivers (e.g. Florida 1996; Noci and Verganti 1999; Cramer 2000). Several companies are striving to stay ahead of legislation and competitors, in order to avoid more or less ad hoc interventions later on (Lamming and Hampson 1996), or to be able to influence future legislation in a way that would give them a competitive advantage (Barrett 1991; Taylor 1992). Expectations of cost savings are also an important factor, and environmentally proactive companies tend to have greater innovative power than other companies (Sharma and Vredenburg 1998; Noci and Verganti 1999).

The growing interest in environmental issues does not only influence the end producers. According to Noci and Verganti (1999) and Hall (2000), awareness and pressure from regulations and customers move upstream along the supply chain and accumulate. Environmental improvements in supply chains are thus attainable through a market-driven process if the end producers include applying environmental performance criteria when selecting suppliers. It is therefore necessary to ask sub-suppliers to meet not only product-oriented purchasing specifications (e.g. cost and quality requirements), but also specifications for environmental performance in the production process (Hall 2000).

To comply with increased requirements from customers and authorities, it is necessary for companies to be aware of the performance of their products throughout their life cycle. One possibility is to measure eco-efficiency in the extended supply chains (ESC). Michelsen et al. (2006) have demonstrated how this approach can be used to compare different products in terms of environmental performance and costs over the life cycle of the products.

The purpose of this paper is to show that eco-efficiency can also be used to assess environmental and value performance when an ESC is redesigned in different ways. This is demonstrated by means of a case study of furniture production. Different scenarios for redesigning the present ESC of a chair have been developed and analysed to quantify the changes in environmental performance within the different scenarios, and their economic consequences.

6.2 Redesigning extended supply chains

When products are analysed to reveal possible eco-efficiency improvements, the extended supply chain should be included. Christopher (1998) defines a supply chain as 'the network of organisations that are involved, through upstream and downstream linkages, in the different processes and activities that produce value in the form of products and services in the hand of the ultimate consumer.' An extended supply chain also includes the use and disposal of the products. The term extended supply chain encompasses both the companies involved and the life cycle perspective. Clift and Wright (2000) and Clift (2003) found significant differences in the ratio between environmental impact and added value in different segments of manufacturing processes. Michelsen et al. (2006) have shown the same for furniture, and revealed that a major part of the environmental impact of the products originated not from the end producer but elsewhere in the ESC. Management of the ESC goes beyond what is normally recognised as supply chain management, as it also includes end-of-life treatment. The ESC is, in principle, infinite, and criteria must be defined for the selection of boundaries. Figure 6.1 shows a simplified picture of the ESC in the present case study, in which the system elements are the components of a chair.

Companies must be able to identify where improvements are possible in the ESC and what impacts these will have on environmental and economic performance. Michelsen et al. (2006) have shown how this could be done by using eco-efficiency. The environmental performance of the ESC is the aggregated environmental impact from all processes in the life cycle of the product, which is assessed using LCA. The value performance of the ESC is the life cycle costs (LCC) of the product, where LCC is defined as the cumulative costs over the life cycle from the users' point of view (cf. IEC 1996). The LCC of a product is thus the price of the product (defined as recommended retail price minus taxes), the average costs in the use phase (cleaning, repair etc.) and the average costs of end-of-life treatment. At present, there is no consensus on how LCC should be defined (Schmidt 2003), but in the present paper, it only includes the actual costs born by the user. This is motivated by the fact that all official bodies in Norway, as in some other countries in Europe, have a legal obligation to take this into consideration when new acquisitions are planned.

When measuring eco-efficiency in ESCs, all scores are compared with a point of reference. This could be an average value for all ESCs that are analysed, or the value for one particular ESC. The data are then presented graphically in XY-diagrams (see Figure 6.2) without merging the value and environmental performances into one single indicator, as is often done in eco-efficiency calculations. This type of data presentation has also been used by others, e.g. in the 'Basel Eco-Controlling Concept' (Schaltegger and Sturm 1998) and at BASF (Saling et al. 2002). If the values are presented as relative values, it is possible to omit everything that is equal in all ESCs and thus simplify the analysis and reduce the uncertainties.

These graphic presentations of eco-efficiency are used to compare different ESCs. However, carrying out improvements requires a more detailed study of the segments in the ESCs. This is done by comparing environmental impact and added costs for the different segments of the ESCs.

Michelsen et al. (2006) used eco-efficiency in ESCs to compare the performance of existing products. However, the same approach can also be used to analyse scenarios in which present ESCs are redesigned to see how this affects their eco-efficiency performance. After a full assessment of a product, different scenarios can be developed, based on the following questions:

- Is it possible to change the materials or the amounts of materials used in the product?
- Is it possible to change the production processes?
- Is it possible to change the product's use?
- Is it possible to change the product's end-of-life treatment?

After potential scenarios for redesign have been identified, these are analysed like any other ESC and compared with the original product. Environmentally and economically viable new solutions are thus identified and the end producer can use this information to redesign the ESC. This does, however, presuppose that they have sufficient power in the supply chain and/or are ready to take responsibility for a larger part of the product's life cycle.

6.3 Case description

The furniture industry is no exception when it comes to the increasing interest in environmental performance. There has particularly been a focus on greater producer responsibility and the possibilities of introducing take-back legislation. In Norway, take-back of furniture was explicitly mentioned in a white paper on environmental policy (Ministry of the Environment 1999). It has also been reported that companies can gain a competitive advantage through their environmental profile (Dahl et al. 2002).

Partly as a consequence of such prospects, furniture industries in several countries have conducted studies to identify opportunities for environmental improvements and evaluate the effects of take-back legislation (e.g. Jaakko Pöyry Infra 2001; Vassbotn and Bjerke 2001; Saft et al. 2003). These studies offer some useful information about ideas prevalent in the industry sector and the findings of preliminary studies, but they were not written in English and as a consequence are poorly accessible.

A paper by Michelsen et al. (2006) compared the eco-efficiency of several chairs designed to be used in conference rooms. The chairs are made by two different manufacturers, and it was found that the flagship model from one of them had the lowest eco-efficiency of all of the models analysed. There was thus an obvious need to improve this model's performance. Therefore we decided to develop different scenarios and assess them to see if it is possible to improve the environmental performance of the chair without increasing the costs. The flagship model has a total weight of 6.81 kg. Table 6.1 shows the main components of the chair. In addition, 3 kg cardboard is used for packaging. Figure 6.1 shows the main components and materials used in the chair.

Table 6.1 Main components of the chair used in the case study

Component	Weight [kg]
Steel frame	1.92 kg
Beech plywood	3.54 kg
Beech	0.44 kg
Polyurethane (PUR)	0.65 kg
Other	0.56 kg

The environmental performance of the ESC was assessed using SimaPro 5.1, selecting Eco-indicator 99 (E)/Europe EI 99 E/E as the impact assessment method. Data on raw materials production were largely based on database values. Transport and energy consumption were included, but waste handling, both by the producer and by suppliers, were included only occasionally. It was assumed that the proportion of recycled steel in the production is 23%. Raw materials for the production of lacquer and plywood adhesive were not included. Nor was the production of raw materials for wool fabrics included, due to lack of appropriate data. Cardboard packaging was assumed to be produced with 100% recycled fibres.

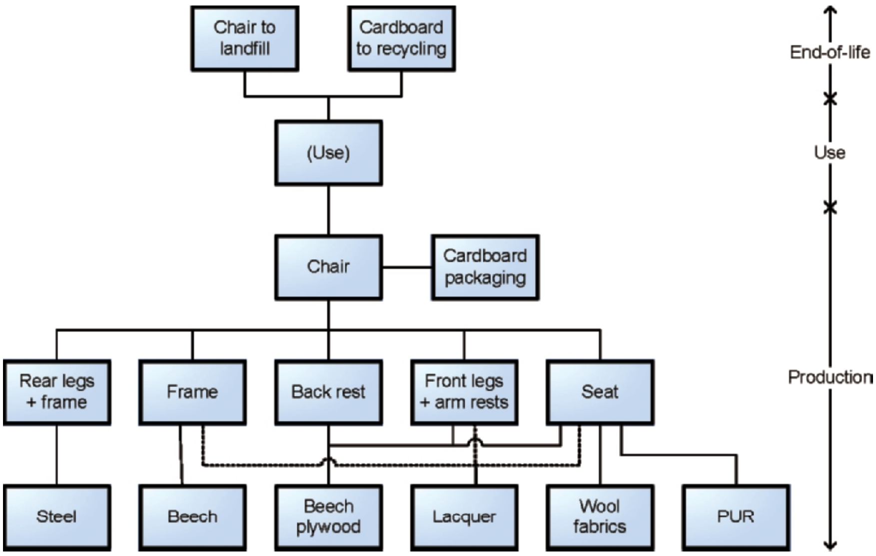

Figure 6.1 Main elements in the extended supply chain of the chair used in the case study

As regards waste handling, database values were used for landfill for all materials except wood. Emission values for wood were taken from Sandgren et al. (1996). According to Vassbotn and Bjerke (2001), landfill is the most likely waste scenario for furniture in Norway.

Land use for transport, beech production or production facilities was not included. In cases where this had been included in database values for different processes, its impact was excluded from the analysis.

In the original case, this yielded an environmental impact of 2030 mPts for the life cycle of the chair. The environmental impact was also calculated with other impact assessment methods (Eco-indicator 99 (H/H), Eco-indicator 99 (I/I), CML 2 baseline 2000 and EPS 2000) integrated in SimaPro 5.1, to check if the choice of impact assessment method had a large impact on the final results.

The life cycle cost is the sum of the price of the product, the expected costs during use and the average costs for disposal or other end-of-life treatment. The producer uses the following equation to calculate the recommended retail price:

$$\frac{(LC + PC) \times 1.15}{0.7} \times k \qquad (6.1)$$

where LC stands for labour costs in production and PC for purchasing costs. This is multiplied by 1.15 to include indirect costs and divided by 0.7 to include the desired margin for the company. The factor k represents the costs and margins for transport and retail. The recommended retail price in 2003 was 2894 Norwegian kroner (NOK)[1].

Costs during use could be related to cleaning and repair. The present case study assumed that there are no costs related to such activities. We also assumed that the chairs are disposed of at a landfill (cf. Vassbotn and Bjerke 2001). In this case, the costs of delivery to a landfill in Oslo were used as disposal costs. At the time of writing, this was NOK 1422 per tonne (taxes not included) (Oslo kommune - Renovasjonsetaten 2004), including transport.

Six different scenarios for changes to the extended supply chain were developed. For the time being, these were limited to changes in materials

[1] 1€ ≈ 7.90 NOK (August 2005)

used (scenarios A-C) and changes to the end-of-life treatment (D-E). Scenarios from these two groups can be combined, as exemplified by one scenario (BE). It is possible to develop scenarios that include alterations to production and assembly processes, but this was beyond the scope of the present study. It was also not considered useful to assess changes in the use of the product, since its contribution to both environmental performance and costs is insignificant (Michelsen et al. 2006).

We did not develop any scenarios that include changes to the amount of plywood, due to the lack of reliable data, especially on the land use impact of forestry. The LCA results indicate that alterations to the wood/plywood content could change the environmental performance significantly. Future work will include the impact of wood components including land use assessment, and a methodology to include land use in forestry is under development (Michelsen 2004).

Scenario A

In this scenario, the use of polyurethane is reduced by 20%. According to the producer of the chair, such a reduction should be possible without reducing the chair's comfort significantly. It is not assumed that this has any impact on the costs, since the reduction will only result in an insignificant decrease in the purchase price of the extruded foam.

Scenario B

In this scenario, polyurethane is partly replaced by an innovative material called Maderon. According to Diaz and Redondo (2002), it is possible to reduce the amount of polyether polyols by 30%, replacing them with cellulose, as well as to reduce the amount of toluene diisocyanate by 35%, replacing it by silicate, in the production of the foam. The environmental performance was estimated based on the alterations to the production phase described by Diaz and Redondo (2002).

The price of the product is not known, but the alteration to the LCC was calculated both on the assumption that the compound is twice as expensive as traditional polyurethane (scenario B) and on the assumption that it is 50% more expensive (scenario B*).

Scenario C

In this scenario, the upholstery is completely omitted. Both polyurethane and fabrics used on the seat are excluded. As a consequence, more lacquer is needed to get an appropriate finish on the seat. The major drawback of this scenario is that it results in reduced comfort and can hence not directly replace the original product.

Scenario D

In this scenario, the chair is dismantled after the use phase. It is assumed that the chair is transported to a dismantling facility close to the user and that this causes no extra emissions from transport and no extra transport costs compared to the present situation (transport to landfill). This could be realistic if the furniture industry had a common dismantling facility and costs and transport due to traditional waste collection were avoided.

It is assumed that the dismantling takes 5 minutes (Vassbotn and Bjerke 2001), and another 5 minutes are added to cover the time used in collection and treatment before the dismantling actually takes place. Labour costs are assumed to be at the same level as those used by the chair's manufacturer. After dismantling, it is assumed that steel is delivered for recycling and the wood for incineration in modern incineration facilities with energy recovery.

We calculated two different cost alternatives. In the first alternative (scenario D), the extra labour costs were included like any other labour cost, as shown in Equation 6.1. In the second alternative, it was assumed that the dismantling would be done as a non-profit activity, with no margin for the dismantler included (scenario D n-p). This was calculated using the following equation:

$$\frac{(LC + PC) \times 1.15}{0.7} \times k + (aLC + aPC) \times 1.15 \qquad (6.2)$$

where aLC stands for the additional labour costs for the dismantling effort and aPC stands for additional purchasing costs (not relevant in this scenario). This presupposes that the work in the dismantling facility is as efficient as that at the end producer's and carries the same level of indirect costs, which again presupposes that large numbers of items are dismantled.

Scenario E

In this scenario, a take-back system is introduced. This scenario assumes that it is possible to collect 80% of the chairs after the use phase. The dismantling time and costs are similar to those in the previous scenario. The cost of the return transport was estimated based on information from Norcargo (2004), on the assumption that 10-20 chairs are transported together. After dismantling, 50% of the steel components are reused in new products, while the rest of the steel is delivered for recycling. Hence, there is an average need for 0.6 steel frames for one new chair, which reduces the purchasing costs and the environmental impact from the production of the steel frames. The rest of the waste treatment takes place according to the original situation.

In the same way as in scenario D, two different cost alternatives were calculated. The first alternative (scenario E) included the extra labour costs like any other labour cost, as shown in Equation 6.1, and extra transport is included as purchasing costs. In the second alternative (scenario E n-p), it was assumed that the dismantling and extra transport is done as a non-profit activity and included as in Equation 6.2.

Scenario BE

This scenario is a combination of scenarios B and E and is thus a scenario where both production and end-of-life treatment are altered. In calculating the LCC, it was assumed that Maderon is twice as expensive as polyurethane. Both cost alternatives from scenario E were included.

6.4 Results

The changes in value and environmental performance for the different scenarios are shown in Table 6.2. The same values are presented graphically in Figure 6.2.

Table 6.2 Changes in environmental and value performances in the scenarios

Scenario	Δ mPt	Δ NOK	Δ NOK (n-p)
A – reduction of PUR	- 30	0	-
B – use of Maderon	- 50	30	B*: 64
C – exclusion of PUR	- 240	- 144	-
D – dismantling and recycling	- 330	30	33
E – take-back and reuse	- 280	4	-142
BE – combination	- 330	24	-12

All scenarios gave an improved environmental performance, ranging from -30 mPts in scenario A to -330 mPts in scenarios D and BE. It is also clear that of these scenarios, alterations to end-of-life treatment had a greater impact on environmental performance than the proposed alterations to the materials used.

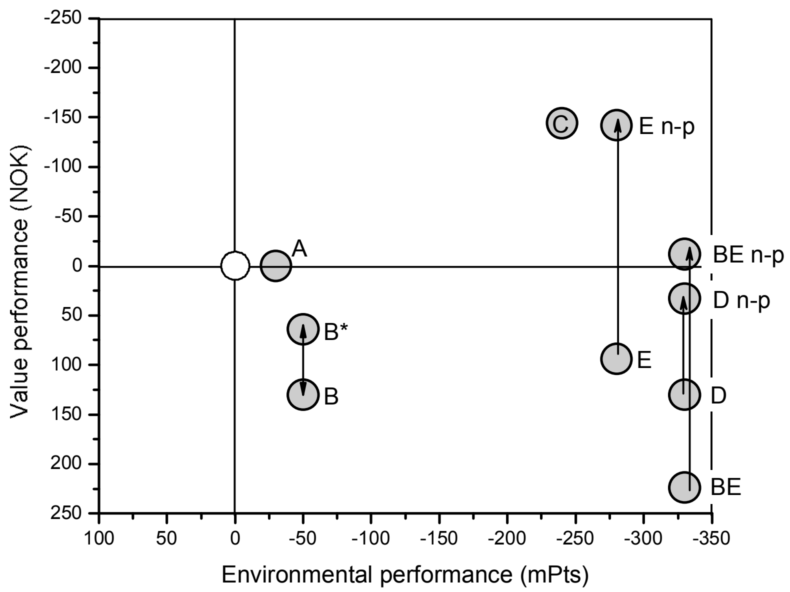

Figure 6.2 Changes in eco-efficiency in the different scenarios (see text for details)

The only scenario giving an unequivocal improvement in value performance was scenario C, which unfortunately involves reduced seating comfort. However, scenarios E and BE also yielded an improved value performance when the dismantling and recycling activities were introduced as non-profit activities.

Table 6.3 Cost-efficiency of environmental improvements in the scenarios

Scenario	NOK/mPt
C – exclusion of PUR	-0.60
E – take-back and reuse (non-profit)	-0.51
BE – combination (non-profit)	-0.04
A – reduction of PUR	-
D – dismantling and recycling (non-profit)	0.10
E – take-back and reuse	0.34
D – dismantling and recycling	0.39
BE – combination	0.68
B – use of Maderon (lower cost alternative)	1.28
B – use of Maderon	2.60

The relative costs of the various alternatives for environmental improvement differed considerably. This is shown in Table 6.3, where positive values indicate the cost in NOK of a reduction in mPts, while a negative value indicates cost reduction. The use of Maderon (B) was by far the most expensive way of improving the environmental performance, even when a lower cost alternative was used. Unsurprisingly, the exclusion of polyurethane and fabrics (C) was the most cost-efficient alternative to improve the environmental performance. Of the scenarios not involving reduced seating comfort, the introduction of a take-back system (E) led to a slightly better performance than dismantling for recovery (D), and as already pointed out, a take-back system also has a potential for cost savings if the extra costs are included as non-profit activities (Equation 6.2).

The picture was more or less the same for the other impact assessment methods we applied. Using EPS 2000 and CML 2, the alterations appeared as greater improvements, giving an environmental impact reduction of more than 24% in scenario D. The only diverging result was that obtained by using Eco-indicator 99 (H). Here, scenarios A, B and C followed the same trend, but scenarios D and E only resulted in about half the reduction of environmental impact compared to scenario C. In addition, scenario E was now slightly better than scenario D.

6.5 Discussion and conclusions

Traditionally, the purpose of eco-efficiency has been to maximise value creation with minimised use of resources and emissions of pollutants (Verfaillie and Bidwell 2000). However, the combination of value and environmental performances in one single indicator has been criticised, since in many cases this obscures conflicting interests with respect to environmental and value performances (e.g. Azapagic and Perdan 2000; Lafferty and Hovden 2002). Alternative solutions with a high eco-efficiency score might simply not be economically viable. This problem is avoided when the eco-efficiency is presented as in Figure 6.2, since both environmental and value performances are presented as they are.

Previous studies have shown that graphic presentations in XY-diagrams are useful for comparing existing products (Schaltegger and Sturm 1998; Saling et al. 2002; Michelsen et al. 2006) and that companies can use the information to evaluate the present performance of their products. The present paper demonstrates the possibility to compare existing products with scenarios for redesigned ESCs. The case study presented above shows the value of expanding the use of eco-efficiency. The results and the way they are presented give companies valuable information in their search for opportunities to improve the ESCs and to assess in what part of the ESCs the improvements should take place.

The results and the graphic presentation are easily understandable for non-specialists. The value performance is expressed as overall costs, which is a familiar measure. No externalities are included. Environmental performance is presented as a single score, which makes it easy to understand even for those unfamiliar with LCA. The graphic presentation clearly visualises which products have the best environmental and value performances. When the graphic presentation is used for different scenarios, as in the above case study, it is also easy to see any improvements. A top-level manager or a purchaser could easily see the range of environmental improvements and the resulting costs or cost reductions.

As in all studies involving LCA, especially those involving comparisons, the quality of the data is critical. In the case study presented here, SimaPro was used to ensure a standardised approach, particularly with respect to normalisation and weighting. However, the use of different impact assessment methods reveals that this actually influences the final results, and there is thus an obvious need for standardised methods within an industry sector if the method used here is to be employed to compare products from different producers (Michelsen et al. 2006). An advantage of the case study presented here is that it used relative values, making it possible to omit data for processes present in all cases. This reduces the uncertainty of the results.

The value performance scores have large uncertainties. We have used the companies' own method of calculating costs, but it is hard to take all eventualities into consideration. The costs of dismantling facilities, for instance, greatly depend on the numbers of items that are dismantled. Costs of reverse logistics are also hardly available. Such costs might be as much as 9 times the costs of delivering the product to the consumer (Persson and Virum 1995), but in scenario E it is assumed that the transport is carried out by a transport company on a case-by-case order. It should hence be possible to reduce the costs in a real situation.

The results of the case study indicate a potential for significant improvements to the current situation, primarily by changing the end-of-life treatment for the chair. While dismantling for recycling yields the greatest environmental improvement, the additional introduction of a take-back system offers opportunities for improved value performance. A take-back system is also a more cost-efficient way of reducing the environmental impacts (Table 6.3). According to Clendenin (1997), Xerox has introduced such systems, for economic reasons. In the case presented here, eventual economic improvements presuppose that extra costs are included as non-profit activities. Clendenin (1997) emphasised the fact that few companies have explored the opportunities for systematic reuse of components, which might explain the apparently low profitability.

Communication with representatives from the industry reveals that there is no common opinion on this subject. There seems to be a tendency for the majority to think that take-back legislation and component reuse is unsuitable, since furniture has a relatively long life expectancy, and models are changed before components are ready for reuse. The idea of component reuse is nevertheless being seriously considered in at least one company.

The results strongly indicate that authorities should consider giving the furniture industry a statutory responsibility for end-of-life treatment. Porter and van der Linde (1995), van den Akker (2000) and Bleischwitz (2003) recommended that authorities should impose requirements for improvements, but that industry should be allowed to find out how to meet them. This is in accordance with the targets for end-of-life treatment for cars, where an EU directive (2000/53/EF) makes no distinction between reuse and recycling. An increased responsibility for the end-of-life treatment also increases the opportunities to address harmful substances. In furniture, this would particularly include brominated flame retardants (Statistics Norway 2003).

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7 Practical experiences with reducing industrial use of water and chemicals in the galvanising industry

Johannes Fresner[a]; Josef Mair[b]; Hans Schnitzer[c], Christoph Brunner[c], Gernot Gwehenberger[d] and Mikko Planasch[d]
[a] *STENUM GmbH, Graz*
[b] *Eloxal Heuberger GmbH, Graz*
[c] *Joanneum Research ForschungsGmbH,*
Institute of Sustainable Techniques and Systems, Graz
[d] *Graz University of Technology - RNS, Graz*

Abstract

While ‘Soft’ factors, like employee training, experience and work instructions can significantly reduce the consumption of water and chemicals by galvanising companies, further significant improvements can be achieved by technical measures. This article demonstrates that the reduction of water and chemicals use can yield significant financial benefits to a company, without compromising product quality or productivity.

Based on the results of a benchmarking survey, a systematic optimisation approach was developed to identify all options that help to minimise water consumption and the use of chemicals, and therefore also sludge generation, while at the same time saving the companies money.

Five case studies identified and implemented measures, including changing the rinsing technology in three pickling plants at the wire producer Joh. Pengg GmbH, the use of spent caustic for neutralisation and an electrolysis plant for copper recovery at the printed circuit board manufacturer AT&S, changing the rinsing technology in the production of printing cylinders by Rotoform and a reorganisation of acid management at the Mosdorfer hot-dip zincing plant.

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All these measures generally reduced wastewater generation by at least 40 % and the amounts of spent process chemicals that have to be treated by half. All measures paid back according to the financial investment standards used by the companies. The paper also discusses the optimal diffusion of this knowledge.

7.1 Galvanic industries and the environment

Galvanic surface treatment is crucial in modern engineering, producing cheap and durable, long-lasting surfaces. Over 10,000 galvanic companies and 8300 so-called in-house galvanics in Europe employ 440,000 people. This number includes the printed circuit board manufacturing industry.

Galvanic processes do, however, cause environmental problems for the companies using them. Galvanic companies generally consume large amounts of water, and the metal salts, acids and caustics applied in the processes have to be removed from the wastewater by expensive treatments before discharge.

About 1 % of the total hazardous waste in Europe is generated by galvanic companies. In 2002, the amount of sludge from galvanic companies in Germany was estimated to be about 80,000 tons annually. Older estimations indicate more than 250,000 t (1997) of sludge without including anodising and pickling companies. About 3 % of the sludges are used as secondary raw materials; the rest is landfilled. According to the Austrian Environmental Agency, about 10,000 t of hazardous and non-hazardous waste result from the Austrian galvanic industry (Sebesta 2002). The Styrian chamber of commerce reports that about 50,000 t of galvanic and hydroxide sludge is produced in Austria annually.

Analyses of companies with similar products show that their water and chemicals consumption varies greatly (Table 7.1).

An Austrian survey in 2001 arrived at similar results (Fresner 2000). Besides the technology employed, organisational factors like dripping time management, staff training or controlling the consumption of chemicals and water had a significant influence on the generation of waste in surface processing companies.

Table 7.1 Specific consumption of degreasing agents, pickling agents and electrolyte in electrochemical zincing in five German companies

	Treated surface area [m²/a]	Pickling agents used [t/a]	Specific consumption pickling agent [t/100,000 m²]	Electrolyte used [t/a]	Specific consumption of electrolyte [t/100,000 m²]	Cleaning agents used [t/a]	Specific consumption of cleaning agents [t/100,000 m²]
1	158,000	24.0	15.0	38.0	24.0	1.2	0.8
2	200,000	202.0	101.0	160.0	80.0	12.8	6.4
3	63,000	21.0	33.0	6.0	9.5	0.1	0.2
4	468,000	150.0	32.0	90.0	192.0	12.4	2.6
5	66,000	1.3	2.0	15.3	23.0	7.0	9.0

It was concluded from these results that there must be a large potential in many companies to avoid and reduce water and chemicals consumption and sludge production, while probably also improving their economic performance by saving expenditure on water and chemicals.

Galvanic companies apply a broad variety of processes, such as degreasing, pickling, etching, passivating, phosphatising, anodising, burnishing, electrophoresis painting, gold plating, silver plating, copper plating, chrome plating, nickel plating, tin plating, zinc plating, etc. The purpose of these processes is to produce a corrosion-resistant metal layer on a base metal or on a plastic surface through chemical or electrochemical treatment.

The parts are first introduced into an acid or alkaline bath to be cleaned, after which the electrolytic or chemical metallisation takes place. After each process bath, the components are cleaned by various techniques, using larger or smaller volumes of water, to avoid impurities being carried over to the next process stage.

The processes differ in the chemical composition of the process baths (degreasing, pickling with caustic soda solution, hydrochloric acid or sulfuric acid, solutions of different metal salts), as well as in their temperature and the use of electric current.

The process water therefore contains, at least to some extent, all process chemicals (degreasing agents, acids, bases, additives or metal ions) and hence has to be pre-treated before being discharged to a wastewater system. Typical wastewater treatment includes neutralisation and precipitation. These processes generate moist sludge, which is usually landfilled.

7.2 The vision of zero emission galvanising

Only integrated and almost closed production processes meet the present legal and economic requirements. Modern galvanic companies use water-efficient methods and reutilise a large part of their metals. As early as 1996, the authors of the 'Rheinland-pfälzischen Branchenkonzeptes' described their vision of 'water-efficient galvanic companies'. 'Process bath constituents are recovered by appropriate procedures and recycled back into the respective process baths.'

Although some German galvanic companies already meet these requirements, it should be remembered that these companies have been newly built, and most were financially supported by public authorities.

Most of the surface treatment plants in Austria are, on average, 10 to 15 years old. These plants face the problem that the process layout is not designed for a water-efficient and zero waste operation.

Obstacles to the revamping of existing plants often include space problems and the uncertainty whether production after the revamping operation will run smoothly and, especially, without quality losses. In addition, each revamping operation usually means a major financial burden to the enterprise, since capital costs are high and production has to be suspended during the re-equipment phase, leading to additional costs. Production stops are a major problem particularly for so-called 'in house' galvanics, since the plants can become bottlenecks to other departments during the reconstruction.

The variety of technologies available for the reduction of wastewater quantities, the recovery of constituents and the maintenance of baths often makes it difficult for companies to select the most appropriate process. Technologies include:

- treatment of process baths with membrane filtration, ion exchangers, electrolysis and thermal processes to achieve a long lifetime for the process baths;

- retention of bath ingredients through appropriate processes like transporting goods with little drag out, splash guards or optimised composition of the baths;
- multiple use of rinsing water through appropriate processes like cascade rinsing or closed water cycles through ion exchangers;
- use of processes for the recovery of raw materials and supplies from process baths or rinsing water (dialysis for nickel, evaporation of chromium, precipitation of zinc);
- substitution of raw materials hazardous to water;
- separate collection and treatment of process wastewater, especially of acid and basic wastewater flows as well as chromium-containing, cyanide-containing, nitrite-containing, precipitating and sulfate-containing wastewater flows.

Experience gained with these measures shows, that improving the ecological situation does not necessarily mean a financial burden. In fact, considering all advantages and savings, ecological measures often lead to economic advantage. The purpose of the ZERMEG project was to help improve the eco-efficiency of enterprises by identifying all measures that are at the same time ecologically and economically efficient and thus motivate companies to implement simple measures which reduce the environmental effects of galvanising.

7.3 ZERMEG: Zero Emission Retrofitting

The ZERMEG approach was developed to address the above problems. The ZERMEG project was carried out within the framework of the Fabrik der Zukunft ('Factory of the Future') programme, and was commissioned by the FFF, the Austrian Research Funds, and BMVIT, the Austrian ministry for innovation and transport. ZERMEG stands for 'Zero emission retrofitting method for existing galvanising plants' (see www.fabrikderzukunft.at). ZERMEG's aim is to define a method to achieve an in-depth analysis of surface processing companies. It provides a guide to help collect data, interpret them and provide ideas for improvement. It also offers a guide for bath maintenance and for closing water cycles, as well as a guide for the implementation of measures to modernise galvanic plants in such a way that

- the amounts of wastewater produced and the pollutants content of the wastewater are minimised;
- constituents of the baths can be recovered;
- non-reusable waste can be recycled by other companies and sectors.

ZERMEG specifically wants to assist in the identification of all measures that have the potential to reduce waste and emissions from a process, and are economically feasible at the same time.

ZERMEG wants to meet these requirements by:

- using a methodical approach;
- providing support by calculation programs;
- providing support by producing reference data and standardised descriptions of technologies;
- offering a discussion platform for an exchange of experiences, for the further development of the model and for the diffusion of data (www.zermeg.net).

7.4 The methodical approach in ZERMEG

The methodical approach divides the analysis into 9 steps: see Table 7.2.

Company analyses in Cleaner Production projects like PREPARE or ECOPROFIT have shown that even in the analysis phase, many effective measures can be implemented with little investment (see: www.prepare.at; www.oekoprofit-graz.at).

Table 7.2 The ZERMEG method

Step description	Activities	Potential for optimisation
1. Analysis of current situation: Measuring water and chemicals consumption	Creating a flow chart, documenting water consumption using a meter, documenting the consumption of chemicals using data from the accounts department, implementing bath-specific documentation of chemicals consumption	Missing data, implementing indicators, daily concentration measurements, avoiding single chemicals where possible
2. Analysis of current situation: Detecting drag out	Measuring, calculating or estimating	Short drain times & exchange times, broad part spectrum, improving the assembly of the parts, base-frame geometry
3. Analysis of current situation: Defining rinsing criteria	Target values from the literature for rinsing criteria and/or conductivity of the last rinsing water; calculating the rinsing criteria used	Quality control, reducing the amount of water, measuring conductivity to control the amount of rinsing water, manually controlling the amount of rinsing water
4. Calculation for comparison: Calculating water consumption	Using the ZEPRA program[a]	Comparing with actual water consumption, volume of rinsing loads, rinsing technologies
5. Calculation for comparison: Calculating chemicals consumption	Using the program	Comparing with actual consumption. If deviations are found: identifying loss flows, technical measures to lengthen lifetimes
6. Defining options for external disposal and recycling	Contact with potential customers and suppliers	Identifying by-products
7. Defining options for possible external recycling	Using the register of water cycle closing technologies	Recycling constituents of baths
8. Evaluating options	Evaluation by financial and sustainable criteria	Pay-back calculations and sustainability evaluation of alternatives
9. Optimising the wastewater plant		

[a] ZEPRA was programmed in MS Excel by DI Gwehenberger (Graz University of Technology - RNS) and DI Christoph Brunner (Joanneum Research, Institute for Sustainable Techniques and Systems).

The internal analysis of the galvanic process improves staff awareness of problems and leads to critical reflection on operational practices. There are a number of important questions the operators have to ask: Are all the additional ingredients actually necessary? What purpose do they have? Do all the baths operate at constant and optimal conditions? Are the rinsing water volumes necessary?

The answers to these questions often yield surprising results. For example, the substitution of some organic bath ingredients can have a positive effect on the organic load of the wastewater as well as on the amounts of galvanic sludge.

All incoming and outgoing material flows should be recorded in as much detail as possible. The following questions should be answered:

- Where are the largest volumes of chemicals in the process?
- Where is water used and how much rinsing water is used?
- When and why are concentrates/semi-concentrates discharged?
- What is the consistency of the data?

The recorded mass flows over a representative period of time are entered into a data sheet. The following data sources are relevant in practice (Table 7.3):

Table 7.3 Data sources for the description of material flows

	Data source
Water	Calculation of water consumption by the accounting department, meters and records
Chemicals	Accounting department, records

Since the correlation between mass flows of chemicals and water and product volumes is also important, the throughput of parts has to be recorded. In order to calculate specific indicators, it is necessary to record the surface areas processed. The surface-related consumption rates of water and chemicals are essential instruments for the identification of measures to reduce consumption.

Additional information must be collected about existing process water flows from the galvanic line to wastewater treatment facilities. These data are the input for the further stages in the process and a reference for later comparisons.

7.5 The ZEPRA program

The ZEPRA program is a tool to minimise the unproductive output of galvanising tanks. It was first developed and tested in a small anodising factory. This computer-based tool had to meet a variety of conditions:

- It had to be applicable to different companies;
- It had to allow quick calculation of variants to the existing process;
- It had to produce a visually appealing output;
- It had to be able to get results without precise knowledge of chemical reactions (working with rules of thumb, experimental data, experience);
- It had to be able to calculate sludge composition to find opportunities for its further use;
- It had to calculate the composition of wastewater to find opportunities for closing water cycles or removing valuable components;
- It had to be easily adaptable to incorporate new knowledge;
- The companies' experts had to be able to use it without intensive training.

Considering these requirements, it was decided to program a new tool using Visual Basic for applications based on Microsoft Corporation's Excel program, rather than using standard software. Standard flow sheeting programs are difficult to integrate with a knowledge base, and standard life cycle analysis software is not suitable to consider recycles and the integration with heuristics regarding chemical reactions.

Excel is a well known spreadsheet program already in use in most companies. The original macro language used in Excel has developed into a programming language that is very similar to BASIC, and can be used for fairly complex programs. Most data can be entered and edited in traditional spreadsheets, while complex calculations are programmed in Visual Basic.

The program represents the flow of materials through the plant, representing the individual process steps by black boxes connected by material flows. One batch corresponds to one program run. It is assumed that the first run of the program uses tanks that have the default concentration supplied by the operator of the plant. This situation, though unrealistic for an existing plant, enabled us to calculate the lifetime of batch units like static rinses. Each additional run uses the concentrations of the previous run, so that after a few runs we get a realistic status of the plant. After each run, the plant status is saved, so it is possible to start at any given point, for example after 100 charges. It is also possible to add or subtract separators for cleansing tanks and/or recycling loops. (See Figure 7.1).

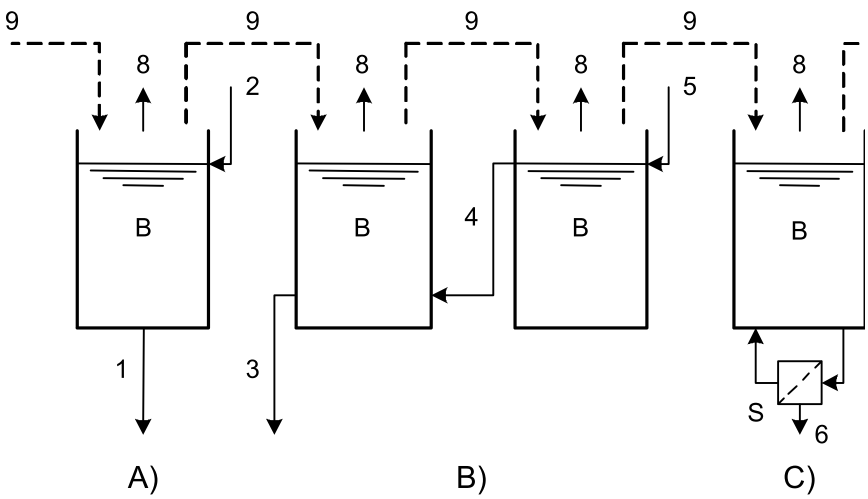

Figure 7.1 Three examples of water flows entering and leaving a bath. All three cases can be calculated with one tool

A) Static tank: batchwise discharge every few hours to every few months; batchwise refill and continuous or discontinuous compensation of evaporation (flow 8).

B) Counter-current rinsing cascade: continuous discharge; overflow from bath to bath; continuous freshwater supply.

C) Bath with some kind of purifying apparatus: continuous or discontinuous wastewater / sludge flow from purifying apparatus (e.g. a membrane filter); continuous or discontinuous compensation of evaporation and compensation of water loss via sludge (flow 6).

A few flows are common to all kinds of arrangements: evaporation, dragout into and out of the baths with the processed items (flows 8 and 9).

Each tank and separator is regarded as one unit and is represented by a single spreadsheet. The flows out of such a unit are the input for other processes. Each unit is regarded as a black box, in that what is happening inside this box does not affect other units, which only see the input and/or output. The output flows are calculated from the input flows in basically three different ways.

The best scenario is one in which all chemical or physical reactions that take place in a tank or separator are known. In this case, it is possible to provide exact output calculations. For most cases, this implies that all the components in the input flows have to been known as well. This is the only way to get theoretical results that are comparable to real measurements. However, even if the main reactions are known, most cases will in fact have to rely on a second-best option.

The second-best option is one in which experimental data of comparable process steps are available, which can be fitted to the actual design of the unit under analysis. In most cases, the results obtained by this way are comparable to results from theoretical models.

Sometimes even the experimental data are missing. In this case, a rule-of-thumb approach must be used, based on the experience of people working at the unit. This includes information like 'we add 50 l of water per day', or 'about a quarter of the chemical is replenished every other week'. These data will yield adequate results for the first assessments, but for further work they will have to be replaced by accurate measurements.

This program is being designed in an open way. The simulation of unit operations that are present in all galvanising plants, like cleaning tanks and rinses, is well advanced. Further unit operations that have already been completed are tanks of anodising plants and steel pickling. However, there are many different galvanising plants and little is known of the chemical processes involved if there are impurities in the chemicals or the water and if the material is a composite alloy of many different metals. As soon as we have experimental data on chemical reactions, they will be incorporated into the program.

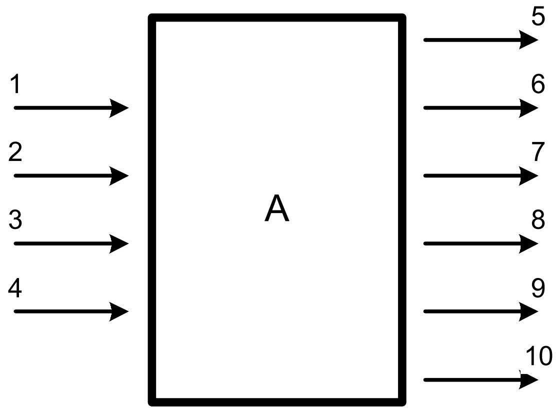

Figure 7.2 Input and output flows per process unit. The black box A is connected to other processes by inflows (1) materials input; (2) water input; (3) dragout input (with material); and (4) recycling stream from process X; and by outflows (5) evaporation; (6) processed materials output; (7) water overflow; (8) dragout; (9) recycling stream to process Y; and (10) sludge/oil/suspended matter

Each process is regarded as a black box A, see Figure 7.2. The processes in this box are chemical, physical or biological reactions. If the detailed reaction is known, the relevant equations are used to calculate the output flows from the input streams. If the exact reactions are unknown, we have to rely on empirical data or our own measurements.

7.6 Using the program

The first step towards improving water efficiency in a galvanising plant involves a thorough analysis of the consumption of water and chemicals in the various process steps. These data can be derived from accounting data, such as the amounts of chemicals bought, or from on-site measurements and asking people who work with the equipment. If the company being analysed has an adequate environmental management system, these figures are relatively easy to find, otherwise it may take a while to obtain the necessary data.

The existing plant is then modelled using the ZEPRA computer program, which automatically generates a speadsheet for each tank. The values of all known input and output flows, the default concentrations of all tanks and other tank-specific values, e.g. rinsing criterion, are entered into the Excel worksheets during the first run, which can be regarded as a setup run. The input to the first tank yields the concentration for the output flows of the first tank. These values are used as input for the second tank and so on, until the preliminary plant has been set up. In subsequent runs, each one representing one lot of material through the plant, the values are calculated using the values of the previous run. All data are saved after each run, so that later sessions can start at a particular state of the plant. After a few runs, it is possible to compare the calculated values with the values obtained by measurement. The calculated values represent the theoretically feasible best consumption values. If he actual values are lower than the calculated ones, one has to assess whether all quality criteria, e.g. rinse criteria, are being met by the current operation in the plant. In most cases, the measured values will be higher, sometimes substantially so, than the calculated ones.

This is followed by the final step, that of interpretation and identification of measures. As a virtual plant has now been created in the computer, parameters can be changed without influencing actual production. It is possible to add or remove tanks, and to add separators to simulate wastewater reuse in other tanks, or add separators to obtain marketable by-products. Some of these alterations simply will not work, but others will, at least in theory. This can be followed up by running traditional experiments in the laboratory to find possible solutions to the problem of reducing the unproductive output of galvanising plants. The main advantage of using this program is that it offers the opportunity to exclude impossible solutions before starting laboratory experiments and perhaps to find unusual solutions not yet tried elsewhere.

The evaluation of the actual situation in terms of plant configuration and material flows should finally lead to the following results:

- transparency of the whole galvanic process, including wastewater treatment, in terms of existing material flows and their relevance to waste generation;
- identifying the main source of relevant material losses;

○ identifying processes with high rinsing requirements;

○ identifying process baths with a high dumping frequency.

Surface-related data about consumption and the concentration of chemicals are important indicators for optimisation. On the one hand, they represent the basic information for daily monitoring, and on the other hand, they provide the basics for the analysis of problems and options for improvement (Fresner et al. 2002). The actual losses due to dragout can be identified by on-site tests in the company, in which a certain number of products or racks are rinsed in a defined rinsing tank, at the usual conditions, and the rinsing water is analysed afterwards.

We regard this approach as a vision-driven approach, which identifies the ideal end result as a starting point for optimisation. This ideal end result is defined by the appropriate rinsing criteria, minimal dragout, optimum useful bath time, economically feasible measures for recycling and maximum external use of spent solutions. This vision can serve as a long-term objective to focus the decisions about possible options for change towards the most useful ones, given the greater picture of the ideal feasible result. This sequence guarantees that the most effective

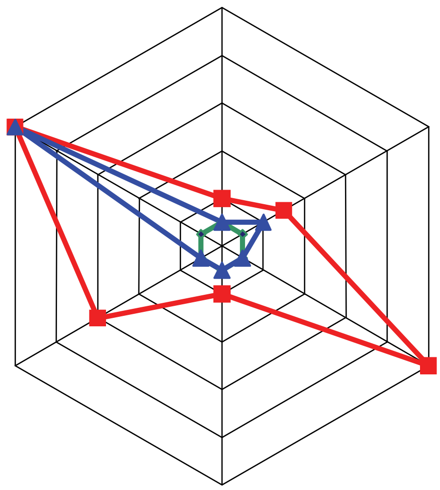

Figure 7.3 This figure shows the ZERMEG Grid for the six basic parameters in the case of the Heuberger anodising plant (see below), to compare the initial situation (outer line) with the results of the optimisation (middle line). The ideal values are represented by the central hexagonal line

measures which at the same time require the smallest investments and operating costs are selected first (Fresner 2004).

We have also developed a tool for the evaluation of the quality of galvanic processes: the ZERMEG Grid. It uses a spider plot with six poles, at which the following parameters are represented by the ratios between their actual values and the ideal values: rinsing criteria, dragout, rinsing water consumption, material losses in pickling, useful lifetime of process solutions and the ratio between external recycling and the reuse of the spent process solutions. The instrument provides ideal values for these parameters (see Figure 7.3).

Since the development of ZERMEG focused on understanding the processes involved in pickling and degreasing various metals, a detailed literature search was undertaken to describe the factors influencing the quality of pickling and degreasing and to prepare models for use in the calculations.

7.7 Case studies

Case study: Eloxal Heuberger

This company with its 24 employees is a typical representative of small to medium-sized Austrian companies. The number of employees has increased from 15 to 24 in recent years, and production has increased from 20,000 m²/year to 90,000 m²/year over the last five years.

The company anodises aluminium surfaces, a process in which the aluminium surface is converted to an oxide film (see Figure 7.4). The resulting dense and hard surface is perfectly connected to the base material. It protects the anodised aluminium to a large extent from corrosion and abrasion. Almost all aluminium alloys can be anodised. The enterprise processes parts of highly diverse origin: profiles for facades, windows and solar plants, sheet metals, but also many small articles (parts for transmissions and engines, exhaust systems, bicycle parts, fine mechanical engineering components and medical technology components).

Since the oxide coatings are heavily dependent on the composition of the material, prior surface treatment must be adjusted to the material to

Figure 7.4 The anodising plant at Eloxieranstalt A. Heuberger GmbH

be treated. This includes an intensive degreasing of the parts, which may enter the factory in an oily or greasy state. Different procedures are used to produce different oxide coatings, which have to meet certain decorative and functional requirements.

The present study determined the average grease and oil film on the materials as they were delivered to the plant by the customers. It was found that burnished and polished parts were practically grease- and oil-free, while mechanically treated parts and sheet metals had an average oil cover of approx. 1 g/m² surface.

After degreasing in a light alkaline bath, the parts are pickled in caustic soda solution to produce a metallic surface. After pickling, the parts are rinsed. This is a critical process, since the adhering pickling solution must be completely and relatively rapidly removed from the work pieces, in order to avoid the so-called 'after-pickling' effect, which would lead to uneven surfaces. Since the pickling solution is very viscous, a relatively thick liquid film sticks to the surface when the parts are taken out of the pickling tank.

We assessed the dragout using more precise measurements. It was shown that the drag out after the pickling were several times higher than was originally expected.

After rinsing, the aluminium parts which have to be processed are immersed in an electrolyte consisting of sulfuric acid. Direct current is then used to produce an oxide coating. Different electrolyte, current and bath parameters yield different characteristics of the resulting surface layer.

After anodising, the parts are again rinsed.

After a final inspection, the parts that are ready for delivery are examined for their technical and decorative quality. The finished parts are carefully packaged.

We introduced detailed records to the plant, which had to be documented on a daily basis: water consumption; energy consumption; chemicals consumption; measurements of the effects of bath concentrations on the basis of daily analyses; discharge rates of the baths; special observations.

This allowed us to point out the following weak points:

The actual dragout – particularly after the pickling – was greater than the theoretical value. This was caused partly by the fact that the pickling solution is very viscous, and partly by metal losses being substantially higher than expected, because some parts were pickled several times to render the surfaces metallically bright again after errors in the subsequent galvanic treatment. Good housekeeping measures (improvement of work instructions and increased control) and intensified control of bath conditions (temperature and chemical concentrations) allowed the error rate to be significantly reduced, from 4 to less than 2% of parts.

With the existing rinsing configuration, the actual amount of rinsing water needed to meet the desired rinsing criterion clearly exceeds the theoretically required value. Concentration profile measurements in the rinses showed that the high density of the media and the partially poorly functioning circulation of the rinses (driven by compressed air) cause a pronounced concentration profile in the rinses[11.] Improved bath circulation by compressed air and the introduction of a constant circulation even during production stops could achieve a clear improvement, in the form of a reduction of the concentration differences.

The anodising baths also showed a significantly increased consumption of sulfuric acid. This proved to be caused by the retardation plant for the extraction of dissolved aluminium. The retardation plant is based on the principle of differential surface absorption of sulfuric acid and aluminium on a resin. This resin was very old at the beginning of the project, and was replaced. This led to a clear improvement of the performance of the plant, and to a significantly reduced acid discharge. These two measures managed to reduce sulfuric acid by more than 30 %.

The program was used to calculate the evaporating water quantities in relation to the temperature and air speeds in the hall. This showed that with the quantities of water used currently, evaporation from the baths is only a small part of the total water consumption. The evaporating quantities could only play an important role in the future if the consumption of fresh water could be clearly reduced.

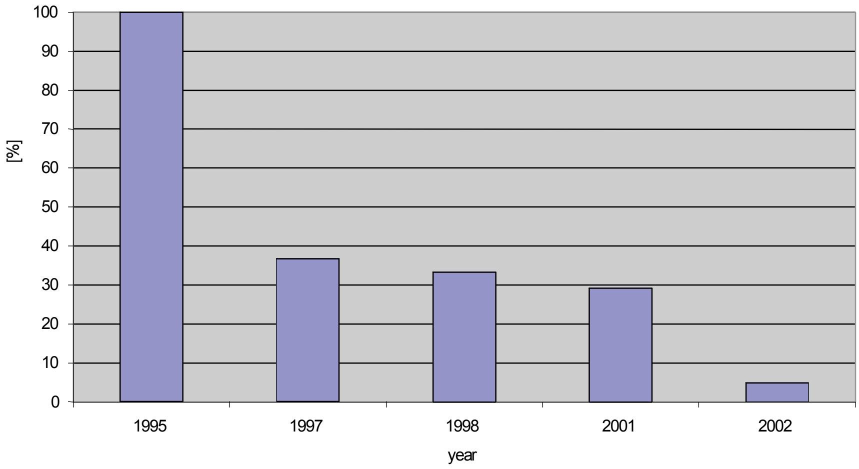

Figure 7.5 Specific water consumption at Eloxal Heuberger (Heuberger GmbH 2002)

Before the study, direct cooling was used for the anodising tank. In order to reduce the huge quantities of cooling water required to keep the temperature of the anodising baths within the required range, a closed cooling cycle with integrated refrigerators was designed. This uncoupled the cooling water quantity from the amount of rinsing water, resulting in an adjustable, tailored amount of rinse water. The investment required for this was approx. 100,000 euros, while the amount saved by reducing water consumption for an annual production of 90,000 m² was approx. 27,000 euros. This calculation does not take the throughput increase and the quality improvement into account. Meanwhile, the specific water consumption could be reduced to less than 40l/m². Some quality problems arose, which were due to the rinsing technology applied. These errors were examined with the help of a metallurgical expert, in order to explain their causes.

We tried to describe and define errors in the starting material, errors in the mechanical pre-treatment and pickling and rinsing errors more precisely, in order to allocate errors. This also served as a basis for additional training activities in the enterprise and considerations for automated control of the water flow through the rinses.

The results show that progress is promising (Figure 7.5). The consumption of acid and caustic solution per treated surface was decreased by around 50 %. This was achieved by the following measures:

- better understanding of the processes in the baths;
- better understanding of the relevant operational sequence;
- better modelling and data collection;
- expertise-building in the enterprise;
- optimising the degreasing tank;
- minimising the metal erosion to achieve the desired effects;
- optimising and considerably extending the lifetime of process baths;
- optimising the use of new technologies for bath maintenance;
- identifying new recycling options.

AT&S

At their plant in Fohnsdorf (Austria), AT&S produce printed circuit boards for cars and mobile phones. As the process plants are changed frequently, the project focused on optimising the wastewater treatment plant. Two options were identified:

- electrolysis of copper to recover copper from concentrates;
- use of caustic stripping solutions for the partial neutralisation of acidic concentrates.

The etching of printed circuit boards generates copper-containing concentrates and rinsing water. Baths of sodium persulfate are used to clean the surfaces and activate them. Depending on the process, an 0.5 to 2 μm layer of copper is dissolved. This copper has to be precipitated in the wastewater treatment plant, and residual persulfate has to be reduced.

An electrolysis plant is an appropriate technology to electrolytically separate the copper from the wastewater. The copper is collected in a very pure form as balls and can then be recycled. At the same time, the persulfate is reduced. The benefits of electrolysis are: minimisation of hydroxide sludge; saving reducing agent; recovery of copper.

A drum electrolysis cell was selected, in which the copper precipitates onto rotating copper balls. The concentration of residual copper is as low as 0.5 to 1.5 g/l. The benefits of this cell are its good hydrodynamic properties and a high electrical efficiency of up to 70 %. The project would not have been feasible if a new plant had had to be bought, but because another plant had shut down, a used, but practically new electrolysis plant with matching capacity could be acquired. The revenue from the recovered copper, the savings on wastewater treatment chemicals and the reduced sludge for disposal resulted in a static payback time of 15 months.

The strongly caustic stripping solutions were purified in the past using filters and an ultrafiltration plant, before being neutralised with hydrochloric acid. In the new situation, the caustic concentrates are used for the partial neutralisation of acidic concentrates in the wastewater treatment plant. Annually, this saves approximately 20 tons of caustic soda and a similar quantity of hydrochloric acid.

Joh. Pengg AG

The firm of Joh. Pengg AG in Thörl (Austria) specialises in the production of wires for sophisticated applications in the automotive, electric and machine manufacturing industries. These products have to have very precise dimensions, and have to meet strict requirements regarding their mechanical parameters. The company is certified according to VDA 6.1, ISO 9001 and QS 9000.

The production of the wires is done in several steps, three of which are galvanising steps. it was these three (batch pickling plant and two continuous pickling plants) which were analysed in this project.

Initially, a wire rod is pickled in a batch plant using hydrochloric acid. After pickling, it is rinsed in a two-step rinsing cascade using cold water. The next step consists of rinsing in hot water in a hot-water tank, followed by phosphatising and rinsing in the hot-water tank. After drying and drawing, the wires are heat-treated, followed by continuous pickling and phosphatising in two plants, depending on the dimension of the final wire. Calculations showed that the consumption of rinsing water could potentially be reduced by up to 80 % in the static pickling. It was decided to change the rinsing technology by combining the two-stage rinsing cascade with the static hot-water rinsing tank to form a three-stage rinsing cascade.

In practice, the volume of rinsing water in the static pickling was reduced by 50 %. As a next step, the rinses in the continuous pickling plants are currently being separated into three-stage rinsing cascades.

In parallel with these studies, a concept was developed in recent months to process the spent acids into a by-product which will be used by another company.

Mosdorfer GmbH

The Austrian hot-dip zincing company Mosdorfer, located in Weiz, produces components for electricity suppliers, and has 30 employees. Mosdorfer is a renowned specialist in this sector. It has developed from a forgery into an innovative partner of companies in the energy, railway and telecommunications sectors.

In the hot-dip zincing plant the following products are processed: components for high- and medium-voltage energy suppliers; insulators; dampers and spirals for electrical installations.

The following baths are used in the zincing plant: degreasing with a mixture of anionic detergents, combined with a continuous filtration with a polypropylene filter medium; four pickling tanks for steel pickling only; two static rinse tank; de-zincing tank; flux tank; drying furnace; zinc tank; quenching tank.

Before, the firm used to buy 150 tons of hydrochloric acid per year. Our calculations showed that this acid consumption could be reduced by 50 %.

The moment at which the acid was dumped in a pickling tank was determined by its zinc contents, as the company had a buyer for the spent acid who demanded a very low zinc content. Zinc was found in significant amounts in all the pickling tanks, because the operators were not careful when selecting a tank for de-zincing defective parts and racks. So a clear separation of pickling and de-zincing became the main goal of the project.

The main weak point was the analytics used to determine the concentration of iron and zinc in the pickling baths. The operators used a graphic procedure based on the density and pH of the samples. This method was unable to differentiate between zinc and iron. Hence, photometric methods were tested for practical application in the company. They also failed, because of the mutual interference between zinc and iron, and between bivalent and trivalent iron ions. A procedure to prepare the samples by extraction was developed in the laboratory. Its implementation in daily practice, however, proved infeasible.

Only analysis by atomic absorption yielded accurate and reliable results. However, sending daily or weekly samples to an external laboratory means a large expenditure for a small company.

In the present situation, the concentration in the tanks is analysed by an external laboratory once a month. The results of these analyses are used to calculate and carefully control the volumes of acids to be topped, so the concentrations of iron and acid are kept within an optimum range.

Today, the de-zincing and pickling processes are scrupulously separated. One acid tank is used exclusively to de-zinc parts for reprocessing and racks. This acid is sold to a company that recovers the zinc from the solution. The separation allows the zinc concentration in the pickling tanks to be kept very low, allowing for a very long useful life of these acids.

As a first result, the specific consumption of hydrochloric acid during the first six month of 2004 was reduced by more than 50 %. The spent acids were completely separated into a fraction rich in zinc and one practically free of zinc. Both fractions are sold as by-products. The zinc-free acid is used in the production of wastewater treatment chemicals.

Rotoform GmbH

Rotoform produces printing cylinders for the graphics industry, and has 20 employees. The firm uses a new standard galvanic line to process the cylinders. After the copper plating or chromium plating, the cylinders run through a surface-processing machine.

The wastewater from the galvanic plants is separated into alkaline and acid flows, then collected and detoxified in a neutralisation plant. A batch treatment is used to reduce the chromium and precipitate the metals. A filter press and an ion exchanger are also used in this process.

The finished cylinders are delivered to printers.

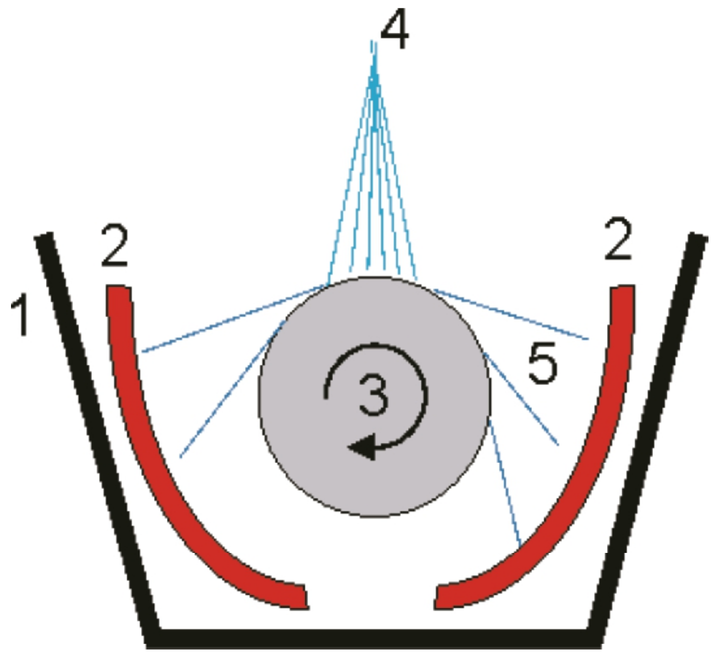

Figure 7.6 Rinsing process in the copper baths. 1. Container of the copper bath; 2. Copper electrode; 3. Rotating pressure cylinder; 4. Spray fog; 5. Water spray on the copper electrodes

The problematic materials in the production of printing cylinders are copper, chromium and nickel. The chromium cycle is largely closed, and the small amount of chromium VI that is lost through the exhaust system is reduced entirely to chromium III in the neutralisation plant.

Via the nickel and copper baths, the respective sulfates enter the neutralisation and wastewater treatment plant. Since it is difficult to break down sulfate, the company is searching for a way to reduce the sulfate contents of the wastewater. The amount of rinsing water per print cylinder was calculated, using equations from the literature to calculate the water consumption of the spray-rinses.

The reason, why significantly larger volumes of water were consumed than theoretically necessary, was the inappropriate geometry of the nozzles and the high water pressure (Figure 7.6), which meant that not only the surfaces of the copper cylinders were sprayed, but a significant portion of the rinsing water was deflected from the surface and rinsed acid from the electrode cage. This effect was minimised by using special flat nozzles and reducing the water pressure. After some test trials with a supplier, an appropriate type of nozzle was found, and was retrofitted in all copper plating machines.

The results were: a reduction of the water consumption of the plant by 40 %, a reduction of the acid dragout by 30 %, and a production increase by 25 %. A further reduction of the consumption of water and acid by 80 % seems feasible in the future.

7.8 Conclusions

The ZERMEG method was applied to five galvanic plants with different processes (wire production, printed circuit board production, hot-dip galvanising, anodising and the production of printing cylinders). The results are very promising.

The rinsing technology used by the wire producer was changed by the following measures: the combination of a two-stage rinsing cascade with a static tank to form a three-stage rinsing cascade and the separation of the rinses in the continuous pickling plants into three-stage rinsing cascades.

The volume of rinsing water in the static pickling has already been reduced by 50 %. At the same time, a theoretical approach that should allow the spent acids to be used in another company has been developed in recent months.

Two improvements were implemented at the printed circuit board manufacturer's: an electrolysis plant to recover copper from etching concentrates and rinsing water; and the use of caustic stripping solutions to neutralise acid concentrates.

This company was able to acquire a practically new used electrolysis plant. The feasibility study showed that the plant should definitely be installed. Because of capacity issues, however, the electrolysis plant was not installed at the location which participated in the project, but at a sister plant, which now recycles 20 kg of copper from the wastewater each day. The wastewater treatment plant now uses caustic concentrates after filtration to neutralise acidic concentrates. This saves 20 tons of caustic soda and a similar volume of hydrochloric acid a year.

At the hot-dip zincing plant, a consistently separated management of pickling tanks was introduced by completely separating the de-zincing and pickling operations. They are currently recycled completely by two other companies. The topping up of the pickling baths is done on the basis of monthly bath analyses and consistent application of the mixing rules. This has reduced the acid consumption in 2004 by 50 % compared to 2003.

In the anodising company, the direct evaporation of the rinsing water offered a good opportunity to install a complete rinsing water cycle. No organic compounds were found in the distillate, and its salts content is very low. This process should be implemented, if there is enough space for a third stage in the two rinsing cascades.

At the printing cylinder manufacturer, the galvanising machines were equipped with new flat nozzles with an optimised geometry, and water pressure was minimised. This reduced the water consumption by 50 % and the acid consumption by 40 %.

We have shown shown that it is feasible to approach the goal of an (almost) zero waste galvanising industry. Important steps towards the realisation of the concept were achieved in the case studies (Table 7.4) by implementing good housekeeping options and simple technology transfer. The implementations included measures which pay back in 0.5–3 years. Additional measures to further decrease the disposal of acids and caustics to the wastewater are technically feasible, but remain too expensive.

The effect of the implemented measures on the environmental impact was assessed using the MIPS model (Material intensity per service unit, www.wuppertalinstitut.de). This model uses standard data to calculate the total resource consumption caused by different materials, including biotic and abiotic resources, water and air. The standard consumption rates, used here for acids, caustic and copper, are given in Table 7.5. The prices used for the evaluation are given in Table 7.6. These prices vary greatly, depending on lot size, availability and local conditions. Table 7.7 presents the results of the environmental and financial evaluations.

The measures at Pengg were implemented to improve the management of the wastewater treatment plant, which is easier at lower hydraulic loads.

Table 7.4 Summary of the ZERMEG results

Company	Reduction of specific water consumption	Reduction of specific consumption of pickling medium (acid, caustic soda)	Other
Anodisieranstalt Heuberger	95 %	50 %	
AT&S	--- [a]	--- [b]	Recovery of 20 kg/d copper, savings of 20 tons/yr of caustic soda, external use of sludge
Joh. Pengg AG	50 %	--- [c]	Complete external use of spent acids planned
Mosdorfer GmbH	--- [b]	50 %	Complete external use of spent acids achieved
Rotoform GmbH	40 %	50 %	

[a] not relevant, because only the wastewater treatment was analysed
[b] no wastewater from rinsing, because rinses are used completely to make up pickling baths
[c] not yet analysed

Table 7.5 Standard MIPS for acids, caustic and copper

Material	MIPS abiotic materials	MIPS water	MIPS air
Copper	179.0	236.0	1.0
Sulfuric acid	0.3	4.0	0.7
Hydrochloric acid	3.0	40.0	0.4
Caustic soda	3.0	90.0	1.0

Table 7.6 Standard prices for chemicals

Material	[€/t]
Copper	3,700
Sulfuric acid	150
Hydrochloric acid	300
Caustic soda	300

Table 7.7 Evaluation of the measures

	Savings		Savings [kg/yr]			Total	Sav-ings	Pay back
Comp	Cate-gory	Total amount	Abiotic MIPS	water MIPS	air MIPS	MIPS	[€]	
Heu-berger	Water	57,000 m³/yr		57,000,000		57,000,000	57,000	<3 years
	Caustic soda	5000 kg/yr	15,000	450,000	5000	470,000	1500	<3 years
AT&S	Copper	4400 kg/yr	787,600	1,038,400	4400	1,830,400	16,280	<2 years
	Caustic soda	20,000 kg/yr	60,000	1,800,000	20,000	1,880,000	6000	<1 year
Pengg	Water	17,600 m³/yr	---	17,600,000	---	17,600,000	---	---
Mos-dorfer	Hydro-chloric acid	75 t/yr	225,000	3,000,000	30,000	3,255,000	22,500	<1 year
Roto-form	Sulfuric acid	50 % (approx.; investment not to be disclosed)						<1 year

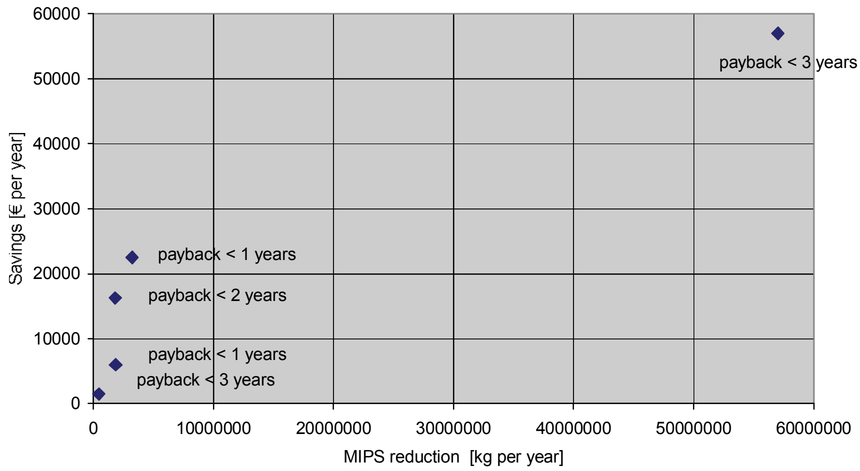

Figure 7.7 Correlation between environmental and financial savings for the measures introduced in the case studies

Figure 7.7 shows a graphic representation of the correlation between the environmental and financial savings for the measures implemented in the case studies. The correlation is positive: the higher the financial savings, the greater the reduction of environmental impact, expressed as total MIPS in kg per year.

The ratio between cost saving and environmental improvement was very similar for most technical improvement options, with one exception. This exception concerned the installation of a cooling plant to reduce the once-through use of cooling water. This improvement alone is achieving an environmental improvement that is nearly ten times larger than that of the other measures together, while the total savings from these options together would be around the same as that from this single superior option. This is due to the huge amount of well water which was previously used at a temperature of 12 °C for direct cooling because of the low temperature required in the galvanising bath (18 °C).

It is concluded that it is possible to achieve a significant reduction in the consumption of water and chemicals and at the same time reduce the generation of sludge to landfill from wastewater treatment, by employing economically favourable measures. This had not been recognised by the companies before the ZERMEG approach was applied. For the range of improvement arrived at during this project, economic and ecological objectives actually converged.

If measures with a pay-back time of more than 3 years have to be implemented, subsidies are needed to reduce the risk for the investor. In view of the present economic conditions, in terms of water and energy prices, this is relevant for most technologies to recycle process baths internally (such as membrane technologies and evaporators).

To disseminate the ZERMEG approach, benchmarks from these applications, documents on the demonstration projects, a manual and a programme for self-analysis of interested companies were made publicly available on www.zermeg.net. This homepage was accessed by 8,500 users in 2004.

Acknowledgements

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8 Cost-efficient solutions can speed up ecological (and social) development – A proposal

Ernst-Josef Spindler
Vinnolit GmbH & Co. KG, Burghausen, Germany

Abstract

The view that positive ecological, economic and social development need to be combined for sustainable development (SD) is a generally accepted concept. In practice, however, the focus is on achieving ecological advantages to support SD, e.g. through IPP (integrated product policy), ecolabelling etc.

This paper shows that integration of economic advantages – by using them sensibly – can achieve huge ecological savings, compared to ecological advantages alone. By way of example, the paper discusses a case in which part or all of the economic advantage of low-cost products is invested in better thermal home insulation, thus saving heating energy. This mainly yields savings of primary energy and various emissions resulting from the burning of non-renewable resources.

The paper formulates a proposal to better support and speed up SD, by using low-cost products and investing part or all of the resulting cost advantage in ecologically sensible optimisations. In all options investigated, this would lead to much greater ecological gain than could be achieved by just purchasing the ecologically most advantageous product. The cost advantage can of course also be invested in social optimisation, such as improving medical services.

The paper concludes that there is no clear relation between ecological and environmental performance. Low-cost products can have excellent results in quantitative life cycle analysis (LCA) and vice versa.

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 211–219.

8.1 Three-pillar model of sustainable development

The three-pillar model of sustainable development (SD) was introduced at the Rio conference and elsewhere. It stresses the importance of developing the ecological, economic and social pillars to overcome fundamental problems like energy resource exhaustion, the greenhouse effect, the increasing gap between first and third world countries or that between the poor and the rich, etc. It has been accepted by all societal groups, including citizens, politicians, NGOs (non governmental organisations), industry etc.
A source of debate between different groups is of course the relative importance of the different pillars. Environmentalists tend to attach the greatest importance to the ecological pillar, while industry might emphasise the economic pillar. This aspect is not discussed here, but results will show the great importance of the economic pillar.

Although many political programmes are based on this three-pillar model of SD, activities seem to be restricted to strengthening the ecological pillar only.

This paper focuses on the economic pillar and especially on the importance of low-cost products and the opportunities they offer to support ecological and social development. The cost advantages of these low-cost products, as derived from economic life cycle cost (LCC) are converted quantitatively into ecological gains, quantified by ecological life cycle analysis (LCA).

The paper partly answers some more general environmental questions:

1. What is the importance of low cost products for SD?
2. How can an industry based on non-renewable resources support SD?
3. Can consumption be in accordance with SD?

8.2 A proposal for supporting SD

People often have to choose between different products, all serving the same needs, all differing in ecological and economic cost, as measured e.g. by LCA and LCC. A very effective strategy to support SD would be to use low-cost products and invest part or all of their cost advantage in optimisation activities which are ecologically and/or socially beneficial.

Examples of ecologically beneficial activities include investment in better thermal home insulation to save heating energy and thus greenhouse emissions, while an example of social benefits would be investment in better medical service. This strategy is very simple, since no profound knowledge of LCA is needed and no LCA has to be applied for specific products. It avoids wasting scarce money on more expensive products and makes money available for optimisation. In many cases, like those discussed in this paper, this strategy leads to huge ecological (and social) improvements. The effectiveness of this proposal can and should of course be verified in doubtful cases.

8.3 The strategy

The following aspects of the strategy are briefly discussed:

- Are other strategies – like eco-efficiency thinking – more effective?
- What is the general importance of cost in SD?
- Does external ecological cost greatly influence LCC?
- Which ecological savings result from a quantitative conversion of economic advantages?
- How can one deal with rebound effects by using the money saved in a positive or negative way?

Role of eco-efficiency in SD

Eco-efficiency is interpreted here as a method to identify optimal ecological, economic (and social) options to satisfy human needs. Continuous improvement of the eco-efficiency of products is a clear target for industry. Some fear, however, that eco-efficiency alone will not be able to solve the ecological problems humankind is facing, and that even the most eco-efficient option can run up against ecological limits, e.g. if it is used by more and more people. Hence, eco-efficiency is a necessary but not a sufficient condition for SD. Additional methods would be needed to deal with increased consumption by growing numbers of people. One of these could be the strategy discussed here, as well as other strategies like sufficiency thinking.

Importance of cost in SD

Cost is an important, perhaps the most important economic parameter. Of course, cost is viewed here from a life cycle perspective, i.e. throughout a product's use time. Low cost is strongly and often positively interconnected with all three pillars of SD:

- Use of low-cost products or product systems saves scarce economic resources.
- Use of low-cost products supports social development, since many people are better able to afford low-cost than high-cost products.
- Use of low-cost products can also support social development, by saving money which can be used for social optimisations, like better medical service.
- Use of low-cost products can support ecological development, since using low-cost products saves money which can be used for ecological optimisations, like better thermal home insulation.

External ecological cost

Discussions of the advantages of low-cost products for SD must include the influence of external cost, since this external cost – if internalised – could cancel out the economic advantage. Only external ecological costs are discussed here, like the cost of CO_2 emissions. The literature provides cost figures for CO_2 emissions ranging from some 4 €/t up to 195 €/t.[1] This large spread in cost figures makes reliable assessment difficult. The spread results from the influence of issues such as economic development, population growth etc., which are beyond the scope of this paper.

Nevertheless, the highest cost figure can serve to evaluate the maximum influence of external CO_2 cost on product cost. The above maximum cost

[1] E.g. European Commission DGXII ExternE Project and others. Highest CO_2-cost e.g. is from Masuhr et al. (1991).

of 195 €/t for CO_2 (Masuhr et al. 1991) is a limit cost required to reduce CO_2 emissions in industrialised countries to 25% of today's emissions. This percentage derives from the target of reducing greenhouse gases by a factor of two worldwide. Industrialised countries need to achieve a greater reduction, i.e. by a factor of four (reduction to 25% of today's emissions) so that greater emission rights can be granted to third-world countries to stimulate their industrial development. In addition, reduction activities will of course become increasingly expensive if the most efficient measures are implemented first. In this sense, 195 €/t is a limit cost for CO_2 to achieve the final and most costly reductions down to 25% of today's CO_2 emissions.

Internalisation of this maximum limit cost for CO_2 of 195 €/t does not greatly influence LCC. By way of an example, let us consider the production of a piping system for drinking water and waste water: the external cost of producing such a system, supplying some 21 houses,[2] would increase the LCC only by some 6.5%, from € 82,163 to € 87,506 (see Table 8.1). If CO_2 emissions are internalised, i.e. converted into monetary value, their LCA result has to be reduced from 27.4 to 0 t (see Table 8.1) to avoid double counting. And since this cost is avoidance cost, the energy demand would also be reduced to around zero. A cost increase of some 6.5% would not be regarded as prohibitive, since competing offers for the work involved would usually differ by much more than 10%.

Other examples, involving windows (Spindler 1999) architectural foils etc, show even smaller increases in LCC. Thus, internalisation of ecological external cost will in many cases not have a great impact on LCC. It should be considered that this internalisation reduces not only CO_2 emissions but also the demand for primary energy (mainly from non-renewable resources) and the emissions of carbon monoxide (CO), nitric, sulfurous acids (NOx, SOx), polycyclic aromatic hydrocarbons (PAH), mercury (Hg), etc.

Table 8.1 shows some LCC and LCA results for the piping system. The results have been averaged over different material options for this system, like PVC (Polyvinylchloride), PE (Polyethylene), cast iron and cement pipes. Cost in Euro (€) and ecological impacts as energy demand in Gigajoules (GJ) and greenhouse gas emissions are shown as negative figures.

[2] See Reuter (1998); parallel to this LCA a LCC study was realised.

The table compares the ‘normal’ piping system with an ‘internalised’ piping system, in which CO_2 is internalised at a maximum CO_2 cost of 195 €/t.

Table 8.1 Some LCC and LCA results for a water piping system

Average piping system	**LCC results**	**LCA results**	
	Cost [€]	**Energy demand [GJ]**	**Greenhouse gas emissions [t CO_2]**
‘Normal’ piping system	- 82,163	- 569.3	- 27.4
‘Internalised’ piping system (195€/t CO_2)	- 87,506	≈ 0	0

Conversion of economic advantage into ecological gain

One efficient energy saving option would be to invest money in better thermal insulation of the walls of the houses. The LCC results presented here derive from an actual sanitation project at a housing estate in Ludwigshafen, Germany.[3] LCA results for thermal insulation were calculated with the help of well-known formulas for heat savings, based on a very efficient heating system. The results of this calculation are shown in Table 8.2, which presents cost and two LCA results for primary energy saved and greenhouse gas emissions saved by investing money in better thermal insulation. The fourth row represents the same situation as row 3, but was calculated for an investment of 1 € instead of investing in 1 m^2 of thermal insulation. LCA results are positive, since they are savings.

The example corresponds to an avoidance cost for CO_2 of 65 €/t, which can be compared with the cost figures dscussed in Section ‘External Ecological Cost’. It is clear that the avoidance cost for CO_2 will increase to some 200 €/t in the future, if the CO_2 emissions are to be reduced to 25% of today’s emissions.

[3] The sanitation project is described in a BASF eco-efficiency study on thermal insulation of houses: www.sustainability.basf.com/de/sustainability/oekoeffizienz/

Table 8.2 Cost and two LCA results for thermal insulation systems

Thermal insulation	LCC result Cost [€]	LCA results Energy saved [GJ]	Greenhouse gas emissions saved [kg CO_2]
1 m^2	- 44.5	+ 11.9	+ 680
0.022 m^2	- 1.0	+ 0.267	+ 15.3

This example can be used to convert the economic advantages of low-cost products into ecological savings. The ecological gain from investing money in better thermal insulation is compared here with the economic and ecological impacts of an average drinking water and wastewater piping system (see Table 8.3). The results show that cost increases of only 2.2% and 2.6% can compensate 100% of, respectively, the greenhouse gas emissions and energy demands associated with the production of an average piping system. Neutralising energy demand (row 'energy neutral pipes') in this case is slightly more expensive than neutralising greenhouse gas emissions only (row 'climate neutral pipes') and more than neutralises the greenhouse gas emissions. In this sense, small economic advantages allow huge ecological optimisations.

Table 8.3 LCC and some LCA results for an average piping system

Average piping system plus thermal insulation		LCC result	LCA results	
Piping system	Plus thermal insulation[m^2]	Cost [€]	Energy demand [GJ]	Greenhouse gas emissions [t CO_2]
'normal pipes'	0	- 82,163	- 569.3	- 27.4
'climate-neutral pipes'	40	- 83,956	- 90	0
'energy-neutral pipes'	48	- 84,292	0	+ 5.1

Positive and negative rebound effects

The cost advantage of low-cost products can be converted into huge ecological optimisations, as shown above. I regard this as a positive rebound effect. Some fear, however, that the money saved could also be used to finance ecologically negative rebound effects.

A positive rebound effect could be the implementation of the optimisation measures shown in Table 8.3, using money to reduce the energy demand or the greenhouse gas emissions required to produce the product.

A negative rebound effect could be to spend the same amount of money to buy gasoline and just burn it for fun. A quantitative comparison shows that one person investing in the positive rebound effect would save as much primary energy and greenhouse gas emissions as eight to ten people would waste investing the same amount of money in the negative rebound effect. Hence, it is more efficient to invest in a positive than in a negative rebound effect.

Optimisation potential and ‘climate-neutral products’

It is clear that investing money in ecological optimisation is only an option. Nobody can be forced to do so. In the following, private and public procurement of goods are distinguished, since they differ in their accountability for the money spent.

Public procurement is accountable to the public for the way money is used, and the public also influences and controls it. In addition, public procurement offers many opportunities to invest money for ecologically useful optimisations, e.g. better thermal insulation for houses owned by a public institution.

Various strategies can be used to prevent private consumers from investing in negative activities and to stimulate them to invest in positive activities. One option would be to make balancing activities available to people who do not have the opportunity to implement special optimisation potentials in their personal environment. This could be achieved by means of a concept like ‘energy-neutral products’ or ‘climate-neutral products’ (see Table 8.3) or ‘carbon-neutral products’.[4] This would be similar to concepts

[4] For more information please contact the author.

like ‘climate-neutral flights’, where a surcharge is paid on a flight ticket by people who wish to balance the ecologically negative CO_2 impact of their flight,[5] or similar to systems in which a higher price for products imported from the third world is used to balance social injustice (‘fair trade’). In many cases, the price increase for ‘energy/climate/carbon-neutral products’ would only be some 1 to 3% of the normal product price.

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[5] For more information: www.myclimate.org

Cases in Products and Consumption

9 Environmental performance of households

Mette Wier[a], Line Block Christoffersen[a], Jesper Munksgaard[a], Trine S. Jensen[b], Ole G. Pedersen[c] and Hans Keiding[d]

[a] *AKF, Institute of Local Government Studies, Copenhagen, Denmark*

[b] *National Environmental Research Institute, Division of Policy Analysis, Roskilde, Denmark*

[c] *Statistics Denmark, Copenhagen, Denmark*

[d] *Institute of Economics, University of Copenhagen*

Abstract

In this study we discuss and apply ways to assess sustainability of household consumption. We propose an integrated environmental-economic model system providing a basis for identifying products or family types representing high environmental pressure. By using this model system, it is possible to develop a general environmental performance index, including weighting the pressure of numerous environmental impacts. We find that middle income families living in single family homes have the least environmentally friendly consumer basket and also constitute a large share of all families in Denmark. Thus, environmental policy measures should be directed towards middle income families living in houses.

9.1 Introduction

Household consumption pattern is a key variable in understanding, assessing and reducing the environmental pressure of the economy. Different family types have different lifestyles, requiring different goods and services, each associated with environmental pressure. Ultimately, all goods are, directly or indirectly, produced for household consumption, either within the producing country or abroad. Since consumption is related to a

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 223–246.

large number of different types of environmental pressure, it is important to include all types of relevant environmental impacts, by e.g. estimating emission profiles. The advantage of emission profiles is that they provide much more information than a single environmental indicator such as energy consumption or CO_2 emissions. The drawback, however, is that large amounts of information may be difficult to interpret. A shift in commodity mix may have multiple effects, e.g., increasing CO_2 emissions and decreasing NH_3 emissions. Due to these multiple effects, it is hard to assess unambiguously whether improved environmental performance is actually achieved or not. Consequently, to reduce complexity, there is a need to weight different types of environmental impacts according to their relative importance, to form a broad environmental performance index, aggregated across environmental impact types.

Weighting together different environmental pressure types requires proper weights. From a neo-classical economic point of view, the ideal environmental index would measure the change in social welfare resulting from the change in environmental pressure. To aggregate across different types of environmental pressure, that is, to weight different types of environmental goods (or 'bads') together, weights should be assigned to each pressure type according to society's preferences. According to neo-classical theory, these preferences, or more specifically, marginal utilities, are revealed by market prices. However, environmental goods are not often supplied and demanded in any market, and hence, have no observable prices. Thus, market imperfections or lack of property rights mean that no such market prices normally exist for environmental pressures[1]. In the economic literature, this kind of market imperfection is described as 'external costs', meaning that environmental damage costs caused by the producer are passed on to society. In Munksgaard et al. (2005), we applied damage cost estimates to weight environmental impacts. For many types of environmental impact, however, damage cost estimates are not available, and this is also the case for some of the impacts considered in the present study. Instead, we suggest an alternative based on a scoring approach in combination with Data Envelopment analysis.

[1] Markets for emission permits have evolved from national targets for the reduction of e.g. CO_2 and SO_2. Reduction targets, however, are not necessarily founded on a strict trade-off between marginal damage costs and marginal abatement costs, as is required in economic theory.

The scoring (or social panel) approach is based on a public opinion poll which, like damage costs (willingness to pay), mirrors individual preferences (Powell et al. 1995). The present study uses opinion poll results on concern about various environmental impact types, which indicate the relative importance (or values) of these effects. Thus, the poll results may be perceived as a rough estimate of the *relative* size of the welfare losses associated with these environmental impacts.

These rough estimates are associated with considerable uncertainty and arbitrariness. To handle this, we apply DEA (Data Envelopment Analysis) to measure environmental performance given a set of restrictions provided by the scoring (public opinion poll) approach. DEA analysis is a non-parametric production frontier approach, measuring environmental efficiency in consumption and production.

Our study distinguishes itself by applying DEA to evaluate the environmental performance of family types, which has not been done before. Furthermore, the study has benefited from recent and detailed data on production, private consumption, environmental impacts and household characteristics.

9.2 The economic model

Input-output analysis is a top-down economic technique, which uses sectoral monetary transactions data to account for the complex interdependencies of industries in modern economies, as well as the flows from industries to final demand categories. Within the scope of life cycle analysis, environmental input-output analysis can calculate factor multipliers, i.e. embodiments of production factors (such as water, labour, energy, resources and pollutants) per unit of final consumption of products produced by industry sectors.[2]

We consider various household types, characterised by a number of socio-demographic variables, and examine their consumption patterns and the associated environmental pressure. Each household has a number of characteristics (family size, age, income, urbanity, etc). Combining information

[2] An introduction to the input–output method and its application to environmental problems can be found in papers by Leontief and Ford (1970) and Proops (1977). The mathematical formalism is described in detail in Lenzen (2001).

about consumption patterns with household characteristics reveals how consumption patterns vary across household types. Thus, each household type can be assigned a bundle of socio-demographic characteristics and demands a bundle of products. For some of these products, environmental pressure arises *directly* from activities taking place within the households, for example, the consumption of fuels and water in the house and the consumption of fuel used to drive a private vehicle. The resources needed and pollutants emitted by households are called *direct environmental impacts*. In addition to these direct effects, we consider environmental pressure arising *indirectly* from private household consumption of products (commodities and services) - that is, the corresponding resources and pollutants needed to satisfy consumer demand. These indirect requirements occur in the numerous industries situated in the countries in which the products demanded by the consumers are produced, as well as in the numerous industries providing raw materials and intermediates. These product flows are modelled by traditional global input-output analysis. The whole process of industrial interdependence is an infinite chain of deliveries, and encompasses domestic as well as imported goods. However, the goods are ultimately being consumed by Danish households. For more on the modelling approach, see Wier et al. (2001).

In this study we use a static input-output model, encompassing direct and indirect emissions embodied in products consumed by households. The direct and indirect emissions of different types from household consumption are estimated using an extended model incorporating emission matrices. For a formal description of the model, see Wier et al. 2005.

9.3 The environmental impact indices

Some emission types are more harmful than others - one kilogram of CH_4 or N_2O contributes much more to global warming than one kilogram of CO_2. Consequently, the emissions are aggregated to an environmental impact index (such as the global warming potential index) according to their potential impact. Each impact index relates to a specific environmental impact type.

We aggregate the various emission types and resource requirements into seven types of environmental impacts according to their relative contributions to a particular type of environmental impact. The environmental indices show the contribution to the various types of environmental impacts: global warming (GWP index), acidification (PAE index), ozone depletion (ODP index), photochemical oxidation (POCP index), air pollution by hazardous substances (APIHS index), water consumption and total material requirement (TMR).

The construction of each of the environmental indices is similar in the sense that the effects of the various substances are calculated as a function of their strength in terms of a common environmental impact type, e.g. global warming potential, acidification potential, ozone depletion potential etc. The intensity with which the individual substances contribute to the environmental impact is, however, different and aggregation thus requires some form of conversion to a common scale before the individual contributions can be added. This is expressed through compound specific effect factors. The effect factors are based on scientific knowledge about substance-specific effects, relating the emissions to environmental impacts. For global warming potential and ozone depletion potential, the indices are, furthermore, based on a uniform scientific consensus (IPCC 1990; WMO 1989) regarding the calculation of the effect factors and the substances included. For more information on the approach, see Appendix I.

9.4 Assigning weights to environmental impacts

As described in the introduction, there is a need to reduce complexity by weighting different types of environmental impacts together in one broad environmental performance index, aggregating across environmental impact types. This weighting should, preferably, be done according to their relative social importance. As noted previously, a way to weight different types of environmental goods (or 'bads') together, would be to assign weights to each pressure type according to societal preferences. One way to measure these preferences is damage cost estimates based on willingness-to-pay estimates. However, damage cost estimates are only available for a limited range of environmental impacts and not for all types of environmental pressure included in our analysis.

Besides damage costs, various other types of weighting are often applied in the field of life cycle analysis. These weights may be based on the distance-to-target approach, environmental control costs, scoring approaches, etc. For a discussion of different approaches, see e.g. Powell et al. (1995) or Udo de Haes (1999). In the present study, we apply a social panel (scoring) approach, namely public opinion poll results. Basically, public opinion polls are, just like willingness-to-pay estimates, a rough measurement of individual preferences (Powell et al. 1995). We use opinion poll results on concern about various environmental impact types, which indicate the relative importance (weights or values) of these effects. Thus, the poll results provide an indication of the relative size of the welfare losses associated with these environmental impacts.[3]

As an illustrative example, we use a EUROBAROMETER survey from 2002, conducted by the European Opinion Research Group on 'The Attitudes of Europeans towards the Environment', which collected results from 309,000 individuals in 15 EU member states[4] (cf. The European Opinion Research Group 2002). Results from this survey, revealing the public's concern about various environmental impacts, are applied in the weighting (for more information, see Appendix II).

These rough estimates are associated with considerable uncertainty and arbitrariness, and it is important to note that the results will be sensitive to the applied weights. A comprehensive analysis of these issues can be found in Munksgaard et al. (2005), where we apply damage costs and perform a sensitivity analysis using the same basic model as in the present paper.

One way to overcome the problem of uncertainty in assigning weights is to apply a non-parametric production frontier approach, viz. Data Envelopment Analysis (DEA). The present study applies DEA to measure environmental performance given a set of restrictions provided by the scoring (public opinion poll) approach.

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[3] Attitudes and concerns stated in opinion polls will be subject to changes over time due to changes in e.g. media attention, political understanding and recognition. Accordingly, the poll results may be perceived as an indication of total societal awareness and valuation at a specific point in time.

[4] We apply opinion poll results for all 15 member states rather than results for the Danish population only, as the activities of Danish households have considerable environmental consequences abroad. This is due to imported goods as well as environmentally harmful substances dispersed, transported and working at a regional and/or global level.

9.5 DEA analysis

DEA was introduced by Charnes et al. (1978) as an alternative approach to the measurement of productivity or technical efficiency in firms or enterprises that are either using or producing goods not directly bought or sold in a market, and which consequently have no observable prices. DEA is a non-parametric method and uses piecewise linear programming to calculate the efficient or best-practice frontier of a sample of units, e.g. producers, countries or products (cf. Charnes et al. 1978). The purpose of DEA is to measure the relative efficiency of each unit considered. The frontier enveloped by efficient units represents the reference technology. Efficiency is normally assessed as the amount of inputs used per unit of output (input productivity) or the amount of output per unit of input (output productivity).

DEA is a useful method when no well-defined theoretical description of the performance of the producers is available. The efficiency scores are calculated relative to an empirically based reference technology. DEA is designed so as to optimise the performance of each individual unit by choosing the best combination of weights for the inputs or outputs related to each unit included in the analysis. The efficiency scores are calculated by comparing productivity ratios (output divided by input), in which the different inputs and outputs are weighted together using weights selected so as to make the relative performance of each unit as favourable as possible. The point is that when the unit comes out inefficient even with the most favourable choice of weights, this does indicate poor performance. For a more formal description of the DEA method, see Wier et al. 2005.

The efficiency of the individual unit relative to this reference technology is calculated in terms of scores on a scale from zero to unity, with frontier units receiving a score of unity. Hence, scores less than unity reflect how much performance should be changed for a unit to become efficient. For efficient units, the score will be one, so that such units cannot be further distinguished with respect to their performance. However, to obtain a ranking of the units according to scores which also comprise the efficient units, one can use the so-called super-efficiency score, which indicates by how much the performance of an efficient unit could be reduced while still remaining efficient compared to other units (Andersen and Petersen 1993).

9.6 DEA in environmental studies

Correspondingly, DEA can be used to compare the environmental performance of units. Using eco-efficiency[5] understood as the lowest possible environmental impact per unit produced or consumed, DEA analysis can provide useful insights into environmental pressure from human activities. The units may be at different levels of analysis, such as firms, plants, industries, products, countries or households. In the present study, we apply the DEA method to weight various environmental impact types together, thus estimating an aggregated environmental performance index (or score) for various household types.

Previous empirical DEA analysis of environmental performance has mainly focused on comparing three types of units: environmental performances of various *countries* have been analysed by e.g. Lovell et al. (1995), Taskin and Zaim (2001) and Zofio and Prieto (2001); those of various sectors, firms, farms or plants by Ball et al. (1994), Bevilacqua and Braglia (2002), Färe et al. (1996), Golany et al. (1994), Hadri and Whittaker (1999), Hailu and Veeman (2001), Jung et al. (2001), Piot-Lepetit et al. (1997), Piot-Lepetit and Vermersch (1998), Reinhard et al. (1999, 2000), Sarkis and Cordeiro (2001) and Tyteca (1997); and finally those of *environmental management systems* by Courcelle et al. (1998), Sarkis (1999) and Sarkis and Weinrach (2001).

Most studies have addressed one single environmental pressure type, most often nitrogen (Ball et al. 1994; Piot-Lepetit et al. 1997; Piot-Lepetit and Vermersch 1998; Reinhard et al. 1999), energy-related emissions such as CO_2, SO2 and NOX (Färe et al. 1996; Golany et al. 1994; Taskin and Zaim,2001; Tyteca 1997; Zofio and Prieto 2001) or waste (Courcelle et al. 1998; Sarkis 1999; Sarkis and Weinrach 2001). Other studies, however, have applied DEA analysis across several pressure (emission) types, e.g. Bevilacqua and Braglia (2002), Hailu and Veeman (2001), Jung et al. (2001), Lovell et al. (1995), Reinhard et al. (2000) and Sarkis and Cordeiro (2001).

Environmental impacts have been treated in various ways: as ordinary outputs, after taking their reciprocal, as undesirable outputs, or as inputs. For a discussion of the strengths and the drawbacks of these approaches,

[5] Note that the concept of eco-efficiency is not unambiguous and is subject to various definitions. For more on this, see e.g. Suh (2004b) or Huppes (2004).

see Dyckhoff & Allen (2001). In this study, we treat environmental impacts as inputs, and consumption (utility) as output, as described above. In so doing, the DEA analysis measures the eco-efficiency per unit of consumption. Thus, the inputs in our model are the seven types of environmental impact indices (based on the total direct and indirect emissions of each unit). Output is consumption (or utility) measured as aggregated value across all goods types. We assume constant returns to scale, meaning that the environmental impact of an extra value unit consumed is independent of the consumption level. This is in accordance with the assumptions in the input-output model system and the calculation of environmental impact indices.

9.7 Imposing restrictions in DEA

DEA can be used without any restrictions on the weights. This is necessary when no a priori knowledge is available. If there is prior knowledge about the relative importance of different inputs or outputs, it is possible to add this information in the form of bounds on the choice of weights. Imposing such restrictions will give better results, and often yield lower scores, since the units are no longer allowed to hide behind favourable but unrealistic weights on particular outputs or inputs where they happen to behave well.

In our case, we do have some a-priori information on the weights, namely the public opinion poll scores described above.[6] To apply this information in the DEA analysis, we impose restrictions on the weights. In so doing, we apply the assurance region approach, introduced by Thompson et al. (1986). Following this method, we impose constraints on the relative magnitude of the weights for the various environmental impact types, using the opinion poll results as weights (i.e. the percentage of respondents stating they are 'very worried' about a given environmental impact). The restrictions are added as pairwise minimum constraints on the

[6] In addition, using totally flexible weights is not always very informative, because the DEA optimisation procedure allows each unit (products or family types) to assign all weight to one environmental pressure type only. Consequently, a very polluting product, performing poorly in relation to all pressure types except one, may turn out to perform very well according to the overall DEA score, because all weight is assigned to the single pressure type for which this product performs very well.

ratio of weights (poll scores) for all effects, making it possible to create a lower bound for each environmental impact type, thus limiting the region of possible weights to a restricted area.[7] Within that area, the weights are optimised using DEA-based eco-efficiency analysis.

9.8 Data

All data used in this study are compatible, meaning that they are based on identical classifications of products and activities. This is an important advantage, as transferring data from one type of classification/aggregation of goods to another causes uncertainty. Thus, it is possible to utilise all the data in an integrated model, using the original data as they are. The data used for the present analysis were provided by Statistics Denmark, and encompass input-output tables, environmental data from the various industries and households, plus family budget statistics. All data sources are linked consistently with the Danish national accounts and input-output tables through common classifications and definitions. Together, the environmental accounts and the input-output tables form a so-called hybrid flow account of the NAMEA type (National Accounting Matrices including Environmental Accounts); see United Nations (2003). More specifically, the following information is included:

- Danish *input-output tables.* These tables comprise 130 industries and 35 categories of final demand (eg. private consumption, public consumption, gross fixed capital formation, exports, etc.). In the most detailed tables, private consumption is further sub-divided into 72 consumption purposes, 5 of which are direct energy consumption by households. The unit of measurement used is DKK 1000s.
- *Accounts for energy flows.* The accounts show balances for the supply and use of 40 types of energy. The usage is allocated across 130 industries as well as households (cf. Pedersen 1999). Energy use is accounted

[7] To illustrate, the weight of Global Warming Potential is restricted to values higher than or equal to 39/29 times the weight of acid rain, where 39 is the percentage of 'very worried' answers regarding climate change and 29 is the percentage of 'very worried' answers regarding acid rain in all EU member states, cf. Appendix II.

for in monetary, physical and calorific terms (TJ). The latter is used in this study.

- *Accounts for water use.* The accounts show the use of groundwater and surface water by 130 industries and households. The unit of measurement is 1000s m^3.
- *Accounts for Total Material Requirement* (TMR). TMR for Denmark is a measure of the global resource extraction necessary to provide the material input to the Danish economy. The first step in calculating TMR is to account for the Direct Material Inputs (DMI). DMI is the sum of the weight of domestic resource extraction and the weight of imports. In a second step, the imports are converted to natural resource equivalents, which express the amount of resources extracted globally in order to produce the imports. The calculation of the TMR includes both used and unused resources. The latter consist of mining overburden, excavated soil for construction, fish discards, etc. The TMR is given in tonnes and allocated across 130 industries and households (cf. Pedersen 2002).
- *Accounts for air emissions.* The following types of air emissions are accounted for: CO_2, SO_2, NO_X, CO, NH_3, N_2O, NMVOC, CH_4, various emission types, i.e. heavy metals, PAHs, halons, HCFCs, PFCs and SF_6. For each type of substance, the accounts show the emissions by 130 industries and households (cf. Pedersen 1999). The measurement unit used is tonnes.
- Family budget data are available from the *consumer survey* by Statistics Denmark (cf. Statistics Denmark 1999). The survey comprises the consumption of 1334 commodities by 3438 representatively selected households. These 1334 commodities are aggregated to the 72 commodities from the input-output tables, and 390 family types can be distinguished in our data. Various household characteristics are recorded, such as number and age of children, number of adults, age of main income provider, type of accommodation, urbanity, socio-economic status and education of main income provider, and type and level of disposable household income and expenditure. Data are collected by recording household purchases on a daily basis, supplemented by personal interviews and information from the registers. The response rate was 68.5%.

9.9 Household environmental performance - results

Results from previous research (surveyed in Wier et al. 2001) addressing energy and CO_2 issues suggest that (1) a substantial part of a household's energy requirements is constituted by non-energy goods, (2) products within the groups of household energy, transport and foods are the most energy- or CO_2-intensive goods, (3) total household energy intensity decreases with income, with expenditure and with urbanity, (4) per capita energy requirements decrease with the number of household members. In the following, we would thus - a priori - expect environmental performance to be better for high income, urban and larger households.

Table 9.1 shows the environmental impact indices for each household (or family) type. Underlying Table 9.1, we can distribute the impacts by 390 family types (i.e. groups of households sharing similar characteristics). For reasons of brevity, however, we show only ten family groups - the top 5 and bottom 5 in terms of environmental performance. Young families are families whose main income provider is younger than 30 years of age. Middle-aged families are families whose main income provider is between 30 and 59 years, and older families are families whose main income provider is 60 years or older. Income brackets are intervals of DKK 200,000.

Table 9.1 shows environmental pressure for the 7 effect types per value unit (DKK 1000) consumed, across ten family types, distinguished by disposable household income per adult in the household, age, house type and urbanity. The last column presents the overall environmental performance indices (or scores) based on the 7 environmental impact indices. Thus, for each family type, the last column shows the environmental performance score (eco-efficiency) relative to household expenditure (per DKK 1000 consumed). Correspondingly, Table 9.2 shows the top and bottom 5 performers in terms of environmental performance relative to household size (per consumer unit).[8]

[8] Consumer units are used as an instrument to control for differences in household size and composition when comparing across households. It is preferable to using the number of persons, as it weights household members according to their decreasing importance in household consumption due to economies of scale and lower consumption by children. In this study, we apply the *modified OECD scale*, in which the first person over 14 years counts as 1 consumer unit, other persons over 14 years count as 0.5 consumer units, and children under 15 years count as 0.3 consumer units.

Table 9.1 Environmental performance relative to household expenditure, top 5 and bottom 5 performers, 1997

	Potential acidification index [mol/1000 DKK]	Global warming potential index [kg/1000 DKK]	Ozone depletion potential index [mg/1000 DKK]	Air pollution by hazardous substances index [g/year*m^3/ng/1000 DKK]	Water consumption [m^3/1000 DKK]	Total material requirement [kg/1000 DKK]	Photo-chemical oxidation index [g/1000 DKK]	Eco-efficiency score *per value unit* [%]
Top 5								
Young, high income, urban	9	66	107	0.52	1.17	286	41	155
Elderly, low income, urban	13	91	90	0.82	1.40	307	65	94
Elderly, middle income, urban	10	79	98	0.45	1.98	289	64	93
Middle-age, low income, urban	12	87	91	0.72	1.36	294	73	91
Young, low income, urban	12	83	98	0.55	1.00	293	72	90
Bottom 5								
Young, middle income, rural	15	110	104	0.44	0.88	338	137	71
Elderly, middle income, rural	14	113	107	0.75	1.25	360	111	71
Middle-age, middle income, rural	15	106	101	0.48	1.02	334	116	75
Elderly, low income, rural	15	126	91	0.70	1.14	339	98	76
Middle-age, high income, rural	14	98	102	0.45	0.93	322	114	77

Table 9.2 Environmental performance relative to household size, top 5 and bottom 5 performers, 1997

	Potential acidification index [mol/1000 DKK]	Global warming potential index [kg/1000 DKK]	Ozone depletion potential index [mg/1000 DKK]	Air pollution by hazardous substances index [g/year*m^3/ ng/1000 DKK]	Water consumption [m3/1000 DKK]	Total material requirement [kg/1000 DKK]	Photochemical oxidation index [g/1000 DKK]	Eco-efficiency score *per consumer unit* [%]
Top 5								
Elderly, low income, urban	13	91	90	0.82	1.40	307	65	126
Young, high income, urban	9	66	107	0.52	1.17	286	41	104
Young, low income, urban	12	83	98	0.55	1.00	293	72	92
Young, low income, urban	13	91	96	0.47	1.00	309	89	91
Middle-age, low income, urban	12	87	91	0.72	1.36	294	73	86
Bottom 5								
Elderly, high income, urban	12	90	108	0.51	0.97	300	92	46
Elderly, high income, rural	14	113	102	0.66	0.67	339	70	46
Young, high income, rural	11	85	115	0.69	0.48	309	97	47
Middle-age, high income, rural	14	98	102	0.45	0.93	322	114	47
Young, high income, urban	13	97	103	0.61	0.86	304	107	52

Table 9.1 reveals that the environmental pressure from each family type (relative to total expenditure value) changes considerably across environmental impact types. A general pattern can, however, be observed for most environmental impact types, as families living in rural houses show the least environmentally friendly performance in all age and income brackets, due to their consumption of energy and transport. It is remarkable that families are rather homogeneous in terms of environmental performance relating to acidification, making differences between the most and least polluting families relatively small.

A look at the overall environmental performance score measuring eco-efficiency per value unit consumed (Table 9.2) reveals that the family type with the best performance is young high income families living in urban blocks of flats. Their score (per DKK 1000 spent) is 155%, meaning that this family type could reduce its environmental pressure by 55%, and still perform best. The second and third best family types are elderly low- and high-income families living in urban blocks of flats. The family types with the lowest scores (per value unit consumed) are young and elderly middle-income families in rural houses. Their score is 71%, meaning that their performance is 71% of that of the best family and that they would have to reduce their environmental pressure by 29% to be as environmentally friendly as the family type having the best performance.

Generally, middle-income families living in houses, especially those living in rural houses, perform worst per DKK 1000 spent. Thus, these families have the least environmentally friendly consumer basket. However, as they do not have the highest expenditure level relative to household size, they perform relatively better in terms of eco-efficiency measured per consumer unit (last column). It is high-income families who perform worst with regard to eco-efficiency relative to household size, simply because they have the highest expenditure level per consumer unit - and not because their consumer basket is less environmentally friendly. In fact, they have the most environmentally friendly consumer basket, as high-income families generally contribute less per DKK 1000 spent.

Looking at contributions to total environmental pressure from the Danish families, the main contributor to all environmental impact types are middle-aged families in the middle income bracket living in urban houses - not only because this family type has a relatively poor consumer basket and a relatively high expenditure level, but primarily because they are the most common family type in Denmark, constituting a very large share (16%) of all family types.

For environmental impact types related to energy and food consumption, environmental pressure per value unit (DKK 1000) spent appears to decrease with income and urbanity, and also to decrease for those living in blocks of flats rather than houses. The main reasons are that energy and transport consumption are lower for families living in flats than for those in houses (especially rural houses), and that energy and foods are necessity goods, meaning that the budget share of these goods relative to total consumption decreases with income. This result holds to some extent for the majority of the other effect types (again related to energy, transport and foods), with the exception of water consumption, ozone depletion and materials consumption.

9.10 Conclusions

Sustainability in household consumption may be assessed using an environmental performance score or index. An overall environmental performance score for households provides a basis for identifying products or family types representing high environmental pressure. However, such indices make it possible not only to evaluate differences across household types, but also to monitor developments and evaluate changes over time. Environmental performance scores offer valuable information to decision-makers as well as to citizens, as they provide a simple way of revealing the success or failure of policies. For environmental policy-making, the relevant question will always be whether a certain change in the environment is good or bad, and also how good or how bad. Environmental performance scores may be a useful instrument to measure the effect and thus to answer such questions. In addition, household indicators reveal the environmental impacts of changes in the composition of family types, thereby assessing the environmental consequences of future demographic and economic scenarios.

In cases where proper weights or values for the environmental impacts are not available, we propose an integrated model system combining a social panel scoring approach in combination with the DEA approach. Using this model system, it is possible to develop a general environmental performance index, including weighting the pressure of numerous environmental impacts. We find that the environmental performance of each family type varies considerably across environmental impact types. The analysis of the overall environmental performance scores shows that families living in urban blocks of flats, especially the young and elderly families, have the most environmentally friendly consumption patterns. Environmental policy measures should be directed towards middle income families living in houses. These families have the least environmentally friendly consumer basket, and also constitute a large share of all families in Denmark.

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Appendix

I. Environmental effect indices and index equations

Environmental index	**Global warming potential index (GWP)**
Index equation	$GWP = \sum_i E_i \times GWP_i \ (kg)$
Index interpretation	Weighting emissions (E_i) in CO_2 equivalence
Substances	i = CO_2, CH_4, N_2O, HFCs, PFCs, SF_6
Effect factor	GWP_i factors (IPCC, 1977; EPA, 2001)
References	IPPC (1990, 1997); EPA (2001)

Environmental index	**Potential acidification equivalence (PAE)**
Index equation	$PAE = \frac{E_{SO_2}}{1/2\, M_{SO_2}} + \frac{E_{NO_x}}{M_{NO_x}} + \frac{E_{NH_3}}{M_{NH_3}} \left(\frac{kg}{kg/mole} \right)$
Index interpretation	Weighting emissions (E_i) in terms of acid equivalents (H^+- ions); (M_i) = molecular weight
Substances	i = SO_2, NO_x, NH_3
Effect factor	
References	Adriaanse (1993), Hauschild (1996)

Environmental index	**Ozone depletion potential index (ODP)**
Index equation	$ODP = \sum P_i \times ODP_i \ (kg)$
Index interpretation	Weighting production volume (P_i) in terms of ozone depleting strength
Substances	i = CFCs, halons
Effect factor	ODP_i factors (EPA, 2001)
References	WMO (1989), EPA (2001)

Environmental index	**Photochemical oxidation potential index (POCP)**
Index equation	$POCP = \sum_i E_i \times POPC_i \ (kg)$
Index interpretation	Weighting emissions (E_i)in terms of an average source specific photochemical ozone formation strength relative to ethylene
Substances	i= NMVOC, CH_4, CO
Effect factor	$POCP_i$ factors (UNECE, 1991, Andersson-Skjöld et al, 1992)
References	Hauschild (1996)

Environmental index	**Air pollution index for hazardous substances- (APIHS)**
Index equation	$APIHS = \sum_i \frac{E_i}{LOEL_i} + \frac{\sum_j E_j \times TEF_j}{LOEL_{benzo(a)pyrene}} \left(\frac{kg}{year} \times \frac{m^3}{g} \right)$
Index interpretation	Weighting emissions (E_i) in terms of a low/no-effect level of human health effects. TEF_j = Toxic equivalence factor relative to benzo(a)pyrene
Substances	i = metals; j = PAHs
Effect factor	$LOEL_i$ = Low effect level values for excess lifetime risk concentration (WHO, 2000); TEF_j = Equivalence factors relative to benzo(a)pyrene (EPA, 1990)
References	Kocher and Greim (2002), Hauschild (1996), WHO (2000), EPA (1990)

Environmental index	**Resource**
Index equation	$\sum$ *water use* (1000 m^3/year)
Index interpretation	Pressure on groundwater resources
Substances	Groundwater
Effect factor	Volume
References	Olsen (2003)

Environmental index	**Resource**
Index equation	TMR (Total Material Requirement) = $\sum$ *natural resources* (kg/year)
Index interpretation	Total weight of materials affected by the national economy
Substances	Direct and hidden input of natural resources
Effect factor	Weight
References	Pedersen (2002)

II. Public opinion poll: Concern about environmental issues

Topics	**Percent in EU15** [a]	**Applied in our analysis as an indicator for the following indices** [b]
Nuclear power and radioactive waste	50	Not applied
Disasters caused by industrial activities	45	Not applied
Air pollution	44	Photochem. Oxidation Index
Natural disasters	43	Not applied
Pollution of tap water	43	Water Consumption
Pollution of underground water	43	
Pollution of seas and coasts	42	Not applied
Pollution of rivers and lakes	42	Not applied
Progressive elimination of tropical rain forests	41	Not applied
Destruction of the ozone layer	39	Ozone Depletion Index
Climate change	39	Global Warming Potential Index
Use of other chemical products	38	Air Pollution by Hazardous Substances Index
Extinction of animal and plant species	37	Not applied
Industrial waste management	37	Not applied
Use of pesticides	36	Not applied
Using up natural resources that cannot be easily replaced	35	Total Material Requirement
Pollution from farming	31	Not applied
Use of genetically modified organisms	30	Not applied
Acid rain	29	Potential Acidif. Index
Domestic and urban waste management	22	Not applied
Urban problems (traffic, public transport, green spaces, etc.)	21	Not applied
Environm. friendly consumption habits	18	Not applied
Noise	18	Not applied
Hunting and shooting	17	Not applied
Damage caused by tourism	17	Not applied

Source: The European Opinion Research Group (2002).
[a] Percent in EU15 is the percentage of 'very worried' answers to the question: 'At present, are you very worried, fairly worried, not very worried or not at all worried about the following topics?' in the EU member states.
[b] Own comment (the authors)

10 Eco-efficiency analysis of an electrochromic smart window prototype

Spiros Papaefthimiou, Elleni Syrrakou and Panayiotis Yianoulis
Solar Energy Laboratory, Physics Department, University of Patras, Greece

Abstract

The environmental efficiency of a prototype electrochromic window was studied using eco-efficiency methodology, combined with life cycle assessment. The data obtained on the specified eco-efficiency indicators provide significant information that could be used in decision-making for the optimisation of the window's energy and environmental performance. The energy efficiency of the product is affected by its life expectancy and the climatic zone. It was found that in cooling-dominated areas the energy needs of buildings can be reduced by more than 55%, while the total energy saved can be 30 times the energy consumed during an expected 25 years life cycle. The corresponding CO_2 and human toxic emissions reductions were estimated to be 6 times those achieved with a conventional double-glazed unit. An expected retail price of 200 euros per m^2 for an electrochromic window would result in a cost of less than 0.10 euros for each kWh saved over a 20-year lifetime. Consequently, purchase cost reduction will be necessary if such devices are to meet market expectations for solar control window products.

10.1 Introduction

Windows incorporating electrochromic (EC) films have been evaluated as a promising subject for research into advanced glazing materials

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 247–259.

(Grandqvist 1995; Karlsson et al. 2001; Sbar et al. 1999; Selkowitz 1994). An EC window is an active solar control device whose transmittance in the visible and near IR parts of the spectrum can be reversibly modulated by applying a low voltage (typically 3–5 V DC). A typical EC device has a 5-layer structure consisting of (1) a transparent and electrically conductive film (ITO) deposited on glass, (2) an electrochromic film (usually tungsten oxide – WO_3), (3) an ion-conducting electrolyte (either a polymer or a solid state compound), (4) an ion storage layer and (5) a second transparent conductive film (Grandqvist 1995). Devices using polymer electrolytes require two glass sheets, one with the ITO/electrochromic oxide and the other with the ITO/ion storage layer. The polymer electrolyte can serve as the laminating medium that holds together the two coated glass sheets. When the external voltage is applied, the lithium ions move into the electrochromic oxide matrix, thus altering its optical properties. The general characteristics, energy benefits and potential savings ensuing from the use of EC windows are summarised in Figure 10.1 (Granqvist 1995; Papaefthimiou et al. 2006; Syrrakou et al. 2005).

An EC window:

- ⇨ has infinite coloration stages
- ⇨ ensures acceptable visual transmittance
- ⇨ can block both direct and diffuse solar radiation
- ⇨ reduces glare and thermal losses
- ⇨ has no maintenance costs (no moving parts)
- ⇨ requires low voltage power supply
- ⇨ provides architects a choice for a "living" building envelope
- ⇨ reduces cooling, heating and ventilating loads
- ⇨ reduces electric lighting
- ⇨ reduces *GHG* emissions

Figure 10.1 Electrochromic window prototype in the coloured state

The system we studied is a 40 × 40 cm EC device, which comprises the five typical layers. Two Pilkington K-Glass™ sheets (called K-Glass hereafter) serve as the transparent conductor. One is coated with the optically active layer, a 350 nm thick WO_3 film deposited by electron gun deposition, while the ion storage layer, a lithium-doped vanadium pentoxide film

($Li_yV_2O_5$) with a nominal thickness of 110 nm is thermally evaporated onto the second K-Glass sheet. The cavity created by these two sheets is then filled with the polymer electrolyte (in a 1.5 mm layer) which is prepared using a solution of a lithium salt mixed with a polymer: lithium perchlorate diluted in propylene carbonate and polymethyl methacrylate ($LiClO_4$ – PC – PMMA). The whole device is peripherally sealed by a silicone sealant to avoid water contamination. A third glass sheet can be integrated (with air or inert gas filling the cavity) to form the final window. The device has luminous transmittance values in the bleached and coloured states of 63% and 2%, respectively, with a coloration efficiency of 50 cm^2/C (see Figure 10.1).

This study used life cycle assessment (LCA) methodology to assess the eco-efficiency performance of the above EC window. The energy efficiency of the device was evaluated by means of environmental performance indicators, to ascertain whether it fulfils its goal as an energy-saving device. Throughout the life cycle analysis we also checked whether it is an environmentally benign product that can really replace conventional glazing to reduce the energy consumed for thermal comfort within a building envelope.

10.2 Methodology

The main objective of the study was to conduct a complete and detailed energy and environmental performance assessment for the prototype EC device. Eco-efficiency is a method to obtain results for both the environmental and economic performances of a product (or process) that can be used in decision-making on investments, product improvement, product selection or public policy. In order to measure and report ecological efficiency, the World Business Council for Sustainable Development recommends the use of indicators based on the balance between economy and environment. Each part of an indicator can be expressed positively or negatively: as value or cost for the economic aspect or as improvement or damage for the environmental aspect. The most appropriate way to define the eco-efficiency indicators is by combining the cost with the environmental improvement and the economic value creation with the environmental damage (International Eco-Efficiency Conference 2004, National

Table 10.1 Eco-efficiency indicators selected for the evaluation of the EC glazing

	Indicator's name	Definition	Units
R_1	Reduction of the building's energy needs	$\frac{\text{Total (heating and cooling) energy savings}}{\text{Single glass energy loads}}$	[%]
R_2	Energy efficiency of production	$\frac{\text{Total energy saved}}{\text{Net energy input}}$	[MJ/MJ]
R_3	Greenhouse gas (GHG) emissions reduction	$\frac{\text{Avoided GHG emissions due to energy savings}}{\text{EC unit}}$	[kg eq. CO_2]
R_4	Human toxic emissions reduction	$\frac{\text{Avoided toxic emissions due to energy savings}}{\text{EC unit}}$	[kg eq. 1,4-DCB]
R_5	Cost intensity	$\frac{\text{Additional purchase cost of EC unit}}{\text{net energy gains (savings - energy input)}}$	[euro cent/MJ]

Round Table 2001). This results in environmental performance indicators based on material and energy balances: raw material used per unit of product (kg/unit); energy used annually per unit of product (MJ/1000L product); energy conserved (MJ); hazardous waste generated per unit of product (kg/unit); emissions of specific pollutants to the air (tonnes CO_2/year) and wastewater discharged per unit of product (1000 L/unit) (see: ISO 1999).

An eco-efficiency analysis of a product would require evaluation of its entire life cycle according to the LCA method. This is an instrument for studying the impacts of a given product over all stages of its life cycle: resource extraction, energy use, production, distribution, use and ultimate disposal (Tyteca 1996). We applied the LCA methodology to the EC glazing we studied, distinguishing four phases according to ISO 14040: (i) goal and scope definition, (ii) inventory analysis, (iii) impact assessment and (iv) interpretation (see: ISO 2000). The goal and scope definition of the LCA provides a description of the product system and its boundaries related to a functional unit. The life cycle inventory analysis (LCI) estimates the consumption of resources (raw materials and energy) and the quantities of waste flows (air, liquid and solid emissions) that are attributable to the product's life cycle. In the next phase of the LCA, the life cycle impact assessment (LCIA) assesses the potential contributions of the inventory data (inputs and outputs) to a number of potential impacts, such as climate change, toxicological stress, eutrophication, etc. The impact categories included in the present study were global warming and human toxicity. The indicators for these two categories, calculated according to the CML guide, were expressed in equivalent kg of CO_2 emissions and equivalent kg of 1,4-DCB emissions per functional unit, respectively (CML 2001).

The LCI analysis and the LCIA yielded the energy data and the results for the impact categories that were needed to calculate the eco-efficiency indicators (SimaPro 2005). These indicators (summarised in Table 10.1) were defined so as to validate the environmental profile of the EC glazing, and are mainly oriented towards the assessment of the energy-saving capabilities and cost-intensity of the system compared to a single–glazed window (the reference case).

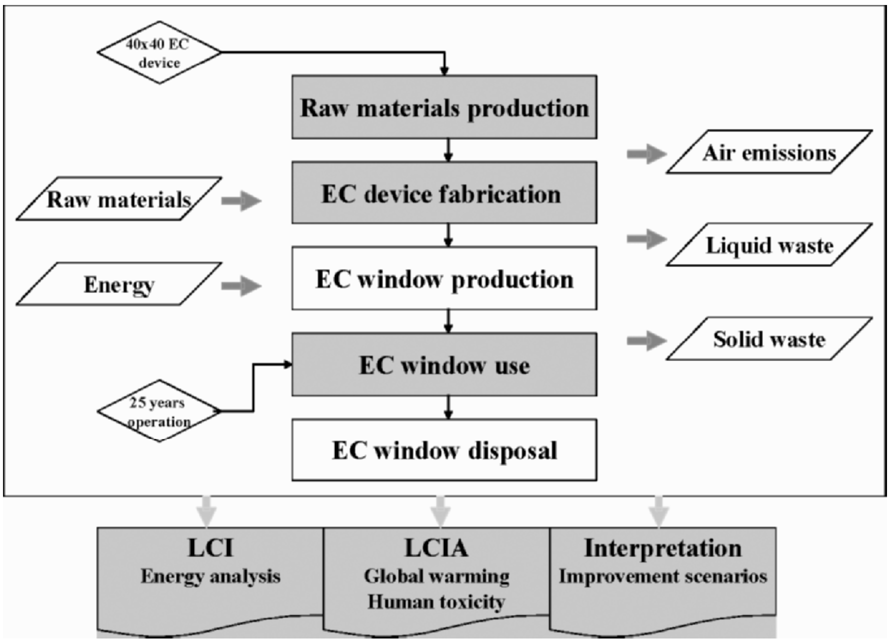

Figure 10.2 System boundaries for the LCA analysis of the EC window

10.3 Life cycle data processing

The functional unit for the LCA study was a typical 40 × 40 cm EC device as described in section 10.1. The system boundaries for the current LCA study are depicted in Figure 10.2, including the phases of production of the raw materials, production of the EC device components, assembly and use. Device disposal and transport were not taken into account. The LCI analysis resulted in a report on the inputs and outputs (energy and substances) for the manufacturing processes and the raw materials production for each part of the device (K-Glass, tungsten oxide, vanadium pentoxide, acetic silicone sealant, lithium perchlorate, propylene carbonate and PMMA; see Syrrakou et al. 2005). In particular, our energy inventory analysis included all the associated energy inputs: the energy embodied in the raw materials production, the energy required for manufacturing and the energy consumed during operation. It was estimated that 180 MJ of primary energy are required for the production of the plain 40 cm × 40 cm EC device prototype, 30% of which is allocated to the primary energy of raw materials and the rest to the electricity consumed during the various manufacturing processes. Only 0.01 kWh are required for the annual operation of the EC unit (Papaefthimiou 2006).

We calculated the reduction of the annual space heating and cooling requirements of a building during the operational lifetime of the device, in order to allow a comparison between the energy spent and saved throughout the anticipated lifetime of the glazing. Our analysis refers to buildings with large facades, located in three characteristic climatic types (cooling-dominated, heating-dominated and moderate areas) which were assumed to replace their single glass windows (SG) with EC glazing. The heating and cooling energy savings were calculated using the RESFEN 3.1 simulation package, for the plain EC glazing and for an alternative type equipped with a spectrally selective coated third glass sheet (EC_{ss}) (see: RESFEN 1999). The control strategy we implemented was based on an evaluation of the desirable ratio between the time that the EC window should be in the coloured and bleached states during the four seasons of the year, finally resulting in the window being maintained mostly in its bleached state during the cold season and in the coloured state during the hot season. The performance of the EC glazing was compared with that of a typical double–glazed unit (DG) and a double–glazed unit equipped with a spectrally selective

Table 10.2 Properties of the glazing types

Window	Description	U-value [W/m^2 K]	SHGC [a] [bl / col]	T_{lum} [b] [bl / col]
SG	1 gl 4mm, clear	5.8	0.74	0.74
EC	K-Glass 4mm / layered EC component / K-Glass 4mm/ air filled gap 9.5 mm / 1 gl 4mm, clear	1.8	0.46/0.10	0.66/0.06
EC_{ss}	K-Glass 4mm / layered EC component / K-Glass 4 mm/ air filled gap 9.5 mm / 1 gl 4mm, spectr. sel. low-e	1.8	0.38/0.10	0.60/0.05
DG	2 gl 4mm, clear, air filled gap 9.5 mm	2.4	0.68	0.67
DG_{ss}	2 gl spectr. sel. low-e, low solar gain 4mm / argon filled gap 13mm	1.40	0.38	0.57

[a] SHGC : solar heat gain coefficient; [b] T_{lum}: luminous transmittance

low-e coating (DG_{ss}), (typically one of the best and most commonly applied solutions in hot climates). The properties of the glazings we studied are listed in Table 10.2, and are based on the assumption that they are operative over a lifetime varying between 10 and 25 years.

In the LCIA stage, we assessed the contribution of the energy savings achieved to the reduction of global warming and human toxicity emissions. We used the electricity energy mix for Greece, where 1.21 kg of equivalent CO_2 emissions result from each kWh used for cooling. As far as the heating energy system is concerned, it has been estimated that 72.3 g of equivalent CO_2 are emitted per MJ of heat produced from natural gas processing (SimaPro 2005).

10.4 Eco-efficiency analysis

The following sections summarise the results of our analysis of the eco-efficiency indicators, comparing the performance of EC with that of other

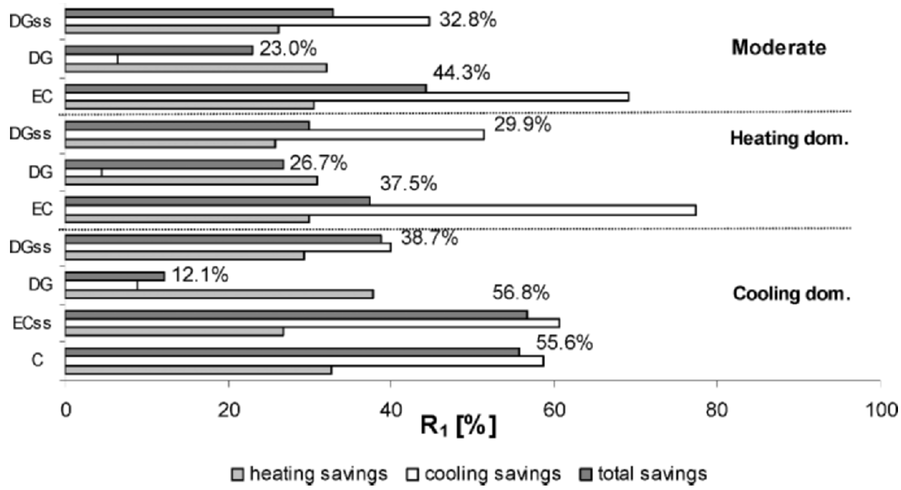

Figure 10.3 Reduction of energy needs of a building R1 with EC and DG glazing

types of glazing. The parameters used for this evaluation were: climatic zone, life expectancy and expected purchase cost.

Reduction of energy needs in buildings

The contribution of the EC glazing to the reduction of the total energy requirements (space heating and cooling) was investigated using the indicator R_1. The use of these fenestration products in cooling-dominated areas can reduce the annual space heating and cooling requirements of a building by more than 50%, as is depicted in Figure 10.3, indicating their high energy performance compared to DG.

The use of EC glazing is to be preferred in cooling-dominated areas, where maximum savings of 5608 MJ per EC unit can be achieved by using EC glazing instead of single-glazed units for a period of 25 years. These savings are attributed to cooling and heating loads reduction, equal to 127.1 kWh/m^2 glass per year and 94.3 MJ/m^2 glass per year, respectively.

Energy efficiency of production

Energy efficiency of production (R_2) is defined as the ratio of energy saved to the total energy used during the life cycle of a device. As depicted in

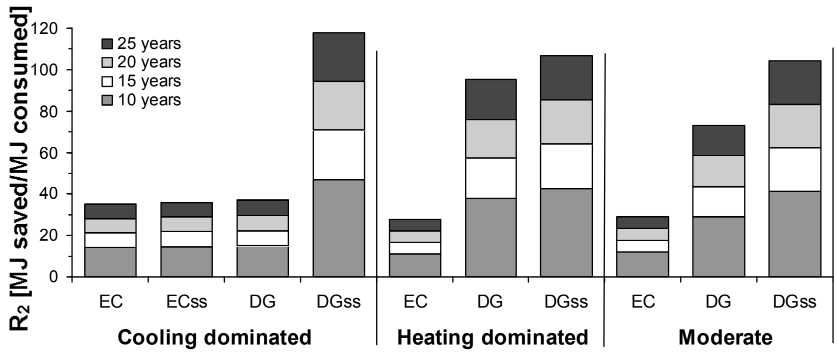

Figure 10.4 Variation in the energy efficiency of production (R_2) for the glazing types studied

Figure 10.4, the energy saved is 30–35 times the energy spent to produce and operate the EC device. The R_2 value for the EC glazing approaches that of the DG only for the cooling-dominated area, while DG_{ss} outperforms the EC in all cases. The reason for this is the fully industrialised production processes for the DG glazing, which reduce the energy input for their production. A possible penetration of EC technology in the glass market will result in incorporation of their production in the online glass industrial processes, thus significantly increasing their energy efficiency of production.

CO_2 and toxic emissions reduction

The reduction of the cooling and heating energy requirements in buildings resulting from the use of advanced glazing provides a strictly environmental benefit: the avoidance of GHG and toxic emissions due to electricity production (used for cooling) and the use of natural gas (for heating). Figure 10.5 depicts the variation in indicators R_3 and R_4 (reduction of GHG emissions and human toxic emissions, respectively) for 25 years of glazing use. The significant contribution of EC glazing to the reduction of GHG and toxic emissions is obvious in the three climatic regions. The emission of more than 600 kg of CO_2 eq. is prevented by the use of EC glazing in cooling-dominated areas. This value is six times larger than the corresponding

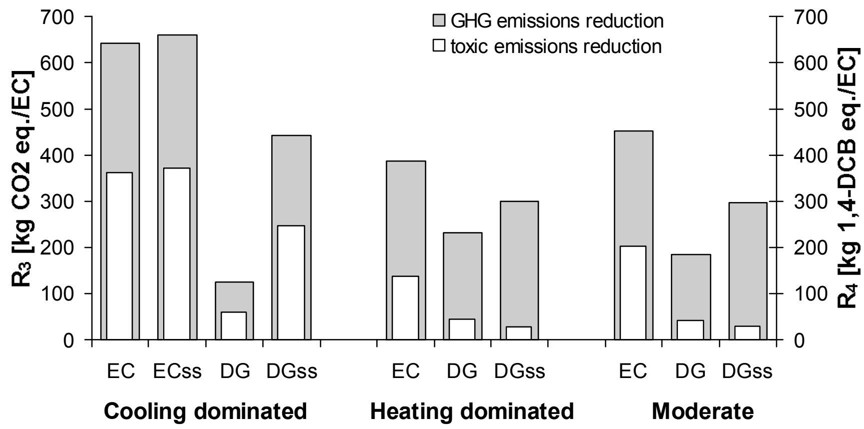

Figure 10.5 Variation in reduction of GHG and human toxic emissions (R_3 and R_4 respectively)

value derived for the use of DG. The corresponding toxic emissions reduction is about 350 kg eq. 1,4-DCB, while the EC_{ss} performs even better.

Cost intensity

Finally, indicator R_5 determines the cost of the net energy gains, expressed in euro cents per MJ of energy gained. The purchase cost of EC glazing is a matter of concern and remains a drawback for its market expansion. A range of prices between 200 and 800 €/m^2 for the EC glazing is examined in Figure 10.6, while DG is currently available at only 30 €/m^2. It is remarkable that increasing the lifetime of the EC glazing to 20 years ensures that the cost per MJ of energy saved is significantly decreased, to less than 1.07 euro cent, which is the current price of electricity in Greece.

10.5 Conclusions

We have evaluated the energy efficiency and environmental impact of an advanced glazing system, consisting of a prototype electrochromic device,

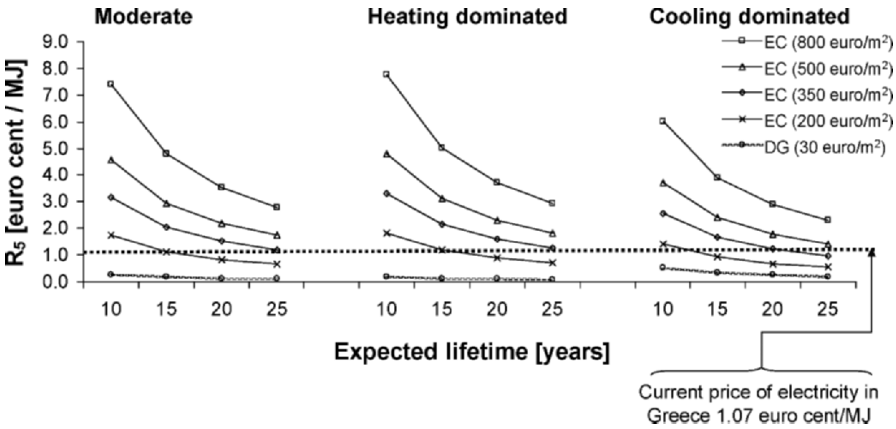

Figure 10.6 Variation in cost intensity (R_5)

employing suitable eco-efficiency indicators and LCA methodology. Such a glazing system can significantly reduce space cooling and heating requirements by replacing the commonly used single-glazed units in buildings, thus acting as an energy-saving component. We selected scenarios for improved control strategy of EC devices implemented in buildings located in three climatic areas, assuming lifetimes ranging between 10 and 25 years. It should be possible to achieve energy savings of 127.1 kWh/m2 glass per year and 94.3 MJ/m^2 glass per year for cooling and heating load, respectively, reducing the energy needs of a building by 55% when the EC device is used in cooling-dominated areas. The corresponding CO_2 emissions reduction is estimated to be more than 600 kg CO_2 equivalent, while the reduction in human toxicity emissions to the air could reach 350 kg 1,4-DCB equivalent, proving the energy benefits and the environmentally benign behaviour of these glazing systems. The energy efficiency of production reaches its maximum value (more than 30 MJ saved per MJ consumed) when the EC glazing is used in cooling-dominated areas and its life cycle is extended to 25 years. Furthermore, if the life cycle is extended to more than 20 years and the device price is reduced to 200 €/m^2, each MJ saved would cost less than the current electricity price (1.07 cent/MJ). This reduction of purchase cost and increased lifetime are the two main targets for achieving both cost and environmental efficiency.

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11 Upgrade planning for upgradeable product design

Kentaro Watanabe[a], Yoshiki Shimomura[a], Akira Matsuda[a], Shinsuke Kondoh[b] and Yasushi Umeda[b]
[a] *Research into Artifacts, Center for Engineering, the University of Tokyo*
[b] *Department of Mechanical Engineering, Tokio Metropolitan University*

Abstract

The current mass production paradigm contributes significantly to environmental degradation. To solve these environmental problems, concepts involving the reuse and remanufacture of products and materials are being proposed. The design of upgradeable products, which could be used for longer than conventional products and encourage people to reuse artefacts, is one of the most promising approaches using these methodologies. In addition, they might provide new business opportunities in the later stages of their product life cycle. Achieving upgradeable design requires a proper plan, which must include information on the upgradeable design, including when a product should be upgraded, with regard to which function, and to what extent. The upgrade plan should also include a solution lineup for upgradeable design that satisfies these conditions. To devise such an upgrade plan, designers need to predict technological trends and user demands. This paper proposes a methodology for upgrade planning based on the prediction of user demand, and on the assumption that technological trends influence user demands. In addition, a methodology is proposed for changing upgrade plans after target products have been distributed, to meet possible fluctuations in technological trends or user demands.

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 261–281.

11.1 Introduction

In order to reduce the environmental problems caused by excessive disposal of products in modern society, products with closed-loop life cycles need to be manufactured. Inverse Manufacturing is one of the most promising concepts to achieve such products (Umeda and Tomiyama 2000). Reuse and remanufacturing, the concepts related to Inverse Manufacturing, are effective means to extend the physical life of a product. However, product life means not only physical life but also value life, which is terminated when functions become insufficient. Many manufactured products, such as mobile phones, are abandoned at the end of their value life. This paper presents a method for extending the value life of products by upgrading them (Shimomura et al. 1999). Efficient upgrading of products requires an upgrade plan which includes the specifications of the products in every product generation. To make this upgrade plan realistic and flexible, it is necessary to predict future technological trends and user demands. Technological trends are assumed to generally influence consumer demands, and on the basis of this assumption, an upgrade plan which includes the prediction of technological trends could be effectively adapted to satisfy user demands. The objective of this paper is to propose a general upgrade planning method. In addition, a method is proposed to modify the upgrade plan to adjust to unpredicted events. Most future events in an upgrade plan can be expressed as changes in the product's components. Therefore, a database should be prepared that can manage the required information, such as product categories, performance and cost, for the purposes of managing and maintaining the information about a product’s structure and components. This information could be used to devise a plan for the design of upgradeable products that could adjust flexibly to future events.

11.2 Upgrade design methodology

Upgrade design is the design of a product so that its performance can be adjusted to changeable user demands by replacing some of its components. Section 11.2 explains the fundamentals of upgrade design and its influence on the environment.

Basic principle of upgrade design

We assume that the performance of a product remains constant until deterioration or breakdown occurs. On the other hand, user demands for product performance generally increase over time, since users become aware of the new functions or better performance of new products in the same category. As time goes on, the gap between their demands and the performance of their own product widens. Thus, it can be assumed that consumers replace their products when the gap becomes wider than a certain threshold level. In such cases, it can be said that the value life of a product has terminated, at least for these users.

Upgradeable design methodology aims to improve product performance to meet user demands by replacing product components and extending the product's value life (see Figure 11. 1).

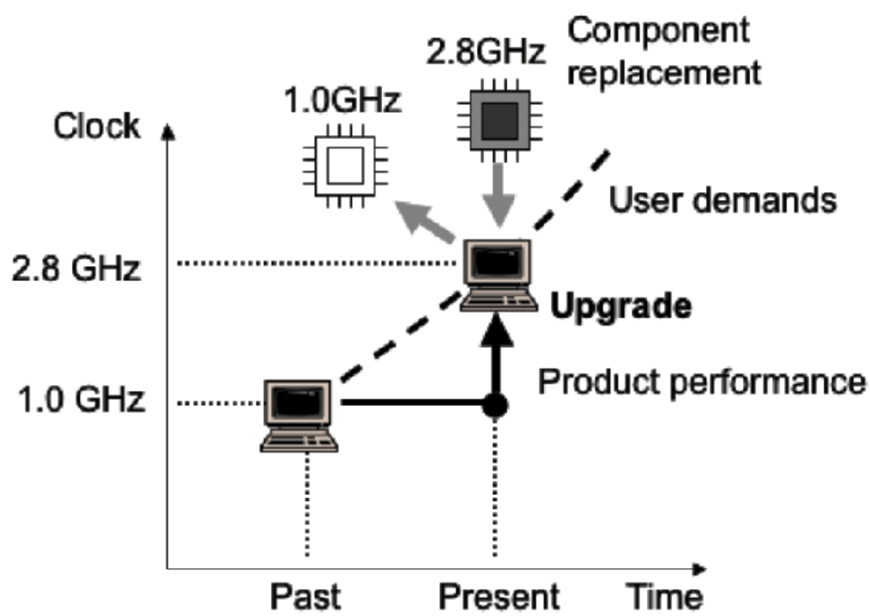


Figure 11.1 Concept of an upgradeable product

Eco-efficiency of upgrade design

Application of upgradeable design reduces the cost of the product's life cycle as well (Ishigami et al. 2002).

(1) Reduction of unnecessary production. The application of upgradeable design results in a reduction of the total amount of production, even though upgrade components must be supplied. This reduces the use of unnecessary materials and energy.

(2) Reduction of waste disposal. In addition to decreasing the number of products used, the replacement of components prolongs the value life of upgradeable products. This should also result in fewer products to be disposed of. Users of upgraded products should enjoy good functionality after component replacement. This important characteristic of upgrade design sets this system apart from other environmentally friendly products or systems.

Upgrade plan

An adequate upgrade of products requires a well-designed upgrade plan that would satisfy the changing user demands. This section explains the structure of such an upgrade plan.

Upgrade design process

Figure 11.2 shows the upgrade design process based on upgrade planning, including two sub-processes:

(1) Upgrade planning. As noted in section 11.1 upgradeable design aims to extend the value life of a target product. To this end, product designers should project future user demands on the target product at the start of the design process. On the basis of this prediction, they must develop a lineup of upgradeable design solutions for a certain period, called an 'upgrade span.' This process allows designers to develop a plan for upgrade design.

(2) Upgrade plan alteration. On the other hand, an unexpected event, called a 'disturbance,' may occur even after the production or sale of an upgradeable product. The designers should respond to such disturbances and change the upgrade plan based on new information. This process allows the upgrade plan to remain adequate.

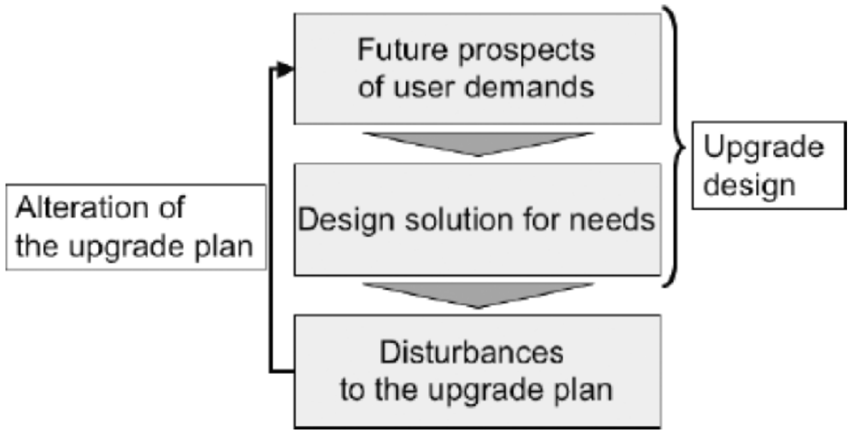


Figure 11.2 Upgrade design process

Design information for an upgrade plan

The upgrade plan contains the following information:

(1) Product performance. Product performance refers to changes in the performance of a target product resulting from technological trends and user demands. Product performance is described by means of parameters.

(2) User demands. User demands are described as parameters of various aspects of the product (such as function and price).

(3) Product structure. This structure is described as a set of components.

(4) Lineup schedule. This is a schedule for introducing a new product, namely a new set of components, to the market. Section 11.4: 'Searching for design solutions', explains how the upgrade schedule is to be determined.

11.3 Description of user demands

In upgrade design, product designers aim to match product performance and user demands. Hence, describing user demands is a very important aspect of upgradeable design. We propose the following two concepts for the description of user demands: (1) Matching user demands and technological trends; (2) Describing the diversity of user demands.

Matching user demands and technological trends

The relationship between a manufacturer of upgradeable products and a user of such products is depicted in Figure 11.3. First, the manufacturer produces products made of existing components. Next, he/she introduces such products into the market. Designers can project the trend of product performance on the basis of a roadmap for the components (see Figure 11.4 (a)). These roadmaps for components and products describe future variations in technological trends, and designers use them to develop an upgradeable product.

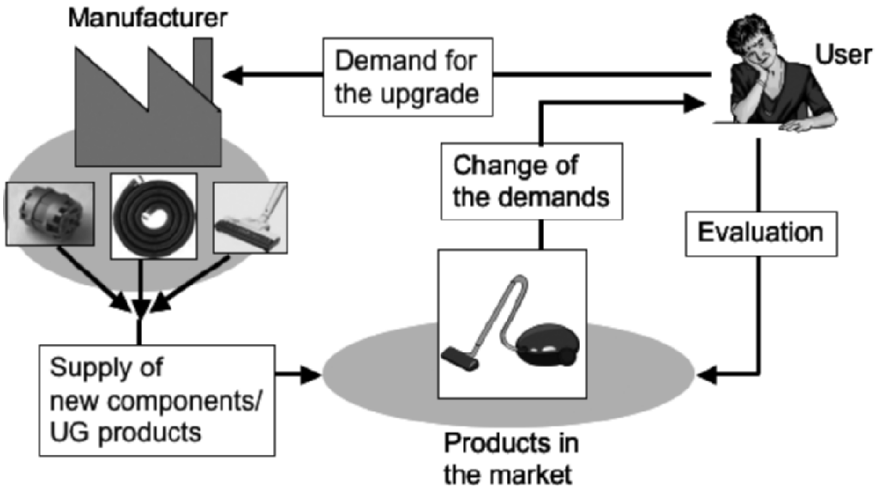


Figure 11.3 Relationship between users and a manufacturer

Consumers then purchase their favourite products after evaluating them on the basis of their own demands. The present study proposes a consumer behaviour model of purchasing products, which assumes that technological trends influence every user demand (see Figure 11.4 (b)). As time goes on, product lineups in the market are refreshed, and some products with higher quality or performance may come into the market. This may cause users to become dissatisfied with their own products. In other words, the gap

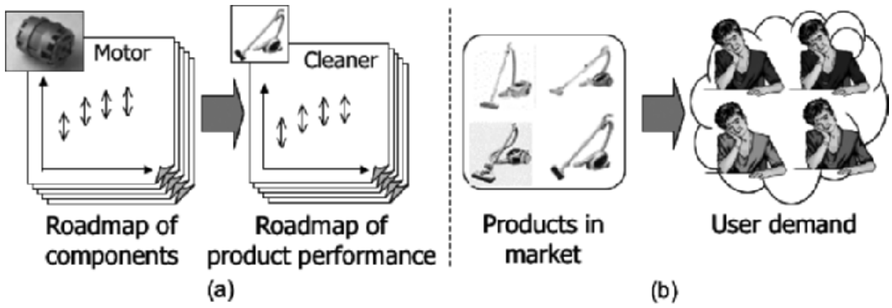


Figure 11.4 Technological trends and user demands

between their demands and the performance of their own product becomes wider. They replace the old product or demand an upgrade from the manufacturer when the gap becomes wider than a certain threshold level.

Describing the diversity of user demands

Since users of any product usually have a large variety of demands, designers have to consider the various demands. We address this issue by categorising user demands as plural patterns and making upgrade plans for every user-demand pattern.

Some patterns of user demands are shown in Figure 11.5. A user with higher threshold levels for product performance is referred to as a 'low-cost user' in this study. By contrast, a high-end user is one with a lower threshold. Using such threshold levels for product evaluation allows various user demands to be expressed. An explanation for the process is presented in 11.4: Setting user-demand patterns.

High-end users	Low-cost users	Professional users
•Demand the highest level of performance •Do not care about costs	•Do not care about performance •Do not buy expensive products	•Demand a high level of performance in some aspects •Do not care about the other aspects of performance •Require the lower price

Figure 11.5 Patterns of user demands

11.4 Upgrade planning

This section focuses on the upgrade planning mentioned in 11.2: Upgrade plan. Product designers can develop an upgrade plan by going through the following process:

(1) Developing a structure model.
(2) Building a component database.
(3) Developing a valuation parameter (VP) roadmap.
(4) Setting user-demand patterns.
(5) Searching for design solutions.
(6) Evaluating the upgrade plan.

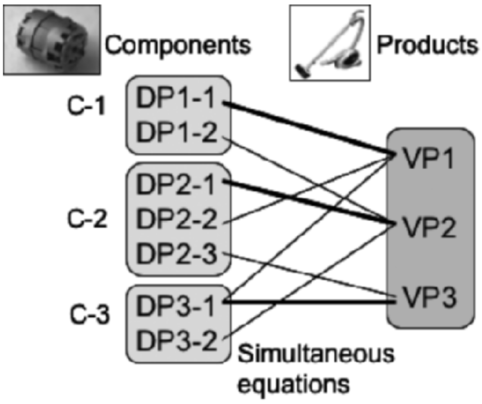
Developing a structure model

Designers first develop a structure model that shows the structural relationships between a product and its components. We assume that the specifications of a product and its components can be described by a set of parameters, which are explained below.

(1) Valuation parameter. When purchasing a product, users should check some parametric values of the product's performance. These may be functional parameters, such as suction power and number of revolutions in a vacuum cleaner, or attribute parameters, such as weight or price. The present study refers to such parameters as valuation parameters (VPs). These parameters can also express the user demands for a target product.

(2) Design parameter. To derive the VPs, a design parameter (DP) was defined to describe the specifications of each component. DPs describe the characteristics of each component, such as electric power and turbine radius (Umemori et al. 2001).

(3) Description of the upgrade. The designers describe the relationship between DPs and a VP of a target product using a set of simultaneous equations (see Figure 11.6), derived from information on the target product structure and its physical characteristics (Ishigami et al. 2002; Umemori et al. 2001). The designers can use the VPs of the target product to follow user demands by determining the values of a DP set and replacing some components. Some components may have strong links with a particular VP. These are the key components of the target VPs.



VP	Key components
1	C-1
2	C-2
3	C-3

Figure 11.6 Structure model

When a key component of a particular VP is not unique, or when one component is the key component of more than one VP, the designers should revise the structure model and solve the side effects caused by replacing the component for the upgrade (Shimomura et al. 1999).

Building a component database

Next, designers research the technological trends relating to the target product. They should examine both the current technology and the future fluctuations in the technological trend during the upgrade span. The result of this analysis is stored in the component database, which contains information about each component of the target product, including the category the product falls under (e.g., motor, fan), a DP set and a schedule for the component to enter and exit the market. A DP set for each component is described as a range of values. The designers express the future fluctuations of each DP as a roadmap on the basis of the information in the component database (see Figure 11.7).

Developing a VP roadmap

The previous step involved developing the DP roadmap, which expresses the fluctuations of future technological trends. In the present step, designers

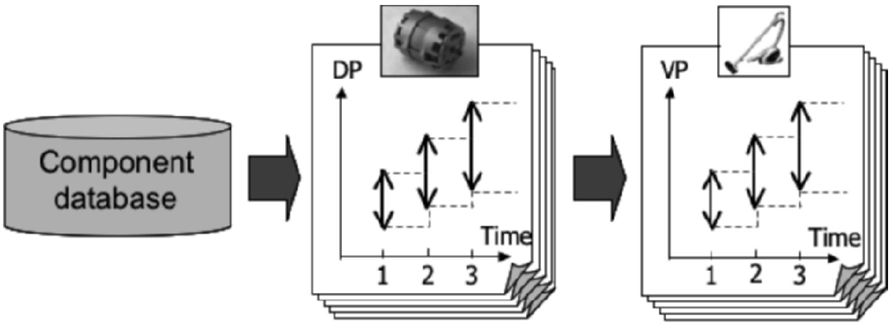


Figure 11.7 DP/VP roadmap

similarly derive the VP roadmap of a target product during the upgrade span.

A VP roadmap can be obtained by applying the simultaneous equations of VPs and DPs determined in the structure model (see Section 11.4 'Developing a structure model - (3)'). This VP roadmap expresses not only the future technological trends but also the upper and lower limits of the future level of user demand.

Setting user-demand patterns

In this part of the process, user-demand patterns are set using the following two operations on the basis of the information on future technological trends in the component database. First, the designers qualitatively classify various user demands as multiple patterns. Second, they divide each VP range into multiple ranges and make them correspond to each user-demand pattern.

Based on the results of these operations, they design the correspondence between user demands and VP range sets (e.g., a user whose user-demand pattern is 'high-end' requires a high-performance product). The present study expresses each user-demand pattern by a range set of VPs.

(1) Qualitative expression of user demands. The designers develop multiple user-demand patterns, such as 'high-end' and 'low-cost' (see Figure 11.5). Each demand pattern is characterised by a qualitative symbol set, such as the three-grade ranking 'H', 'M' and 'L'. Table 11.1 summarises

the user-demand patterns for a vacuum cleaner. For example, the high-end user presented in Table 11.1 requires a vacuum cleaner with suction power and clean emissions in the 'H' rank and noise in the 'L' rank.

(2) Quantification of user demands. Based on the VP roadmap and several user-demand patterns, the designers express the fluctuations in a VP range set for every user-demand pattern as time-series. Table 11.2 describes the correspondence between three user-demand patterns for the VP 'suction power of a vacuum cleaner' and the qualitative ranking and quantitative VP range set.

Designers can express user demands as a set of concrete and quantitative value ranges by describing the time-series information of a user-demand pattern. For example, as shown in Figure 11.8, a CPU with a speed of 2.4 GHz has been displaced from the 'H' rank at T_2 because of shifts in user demands. This time series behaviour of VP ranges can be used to determine a target value for the product upgrade.

Searching for design solutions

In this step, designers search for design solution lineups to satisfy each user-demand pattern. The VP range set of the quantified user-demand patterns is the target value of the design solution in each generation.

Table 11.1 Qualitative user-demand pattern

VP name / User demands	Suction	Noise	Emission
High-end	H	L	H
Low-cost	L	H	L
Professional	M	M	H

Table 11.2 Relationship between VP ranks and VP ranges

Suction user demands	VP rank	VP range [W]
High-end	H	[600-480]
Low-cost	L	[600-180]
Professional	M	[600-360]

Speed / Time	VP rank	VP range [GHz]
T_1 (Past)	H	2.4
T_2 (Present)	M	

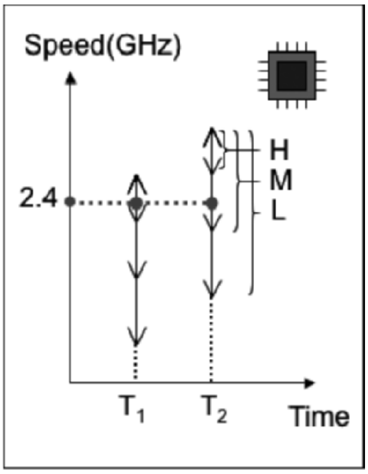


Figure 11.8 Shifts in VP rank and VP range

Process before the first upgrade. First, designers tentatively choose a design solution for the first generation. This solution may be modified afterwards. Next, they examine the basic specifications of a design solution for the next generation. This specification is derived from the design solution for the first generation. It contains the target VPs to be upgraded, a timing of the upgrade, and a set of VPs.

Timing of the upgrade. The product upgrade should be effected when any VP is displaced beyond the range of user demands. Figure 11.9(a) depicts two VP roadmaps (VP1, VP2) for a vacuum cleaner. In this case, VP1 is the first to exceed the range of user demands. Designers should perform an

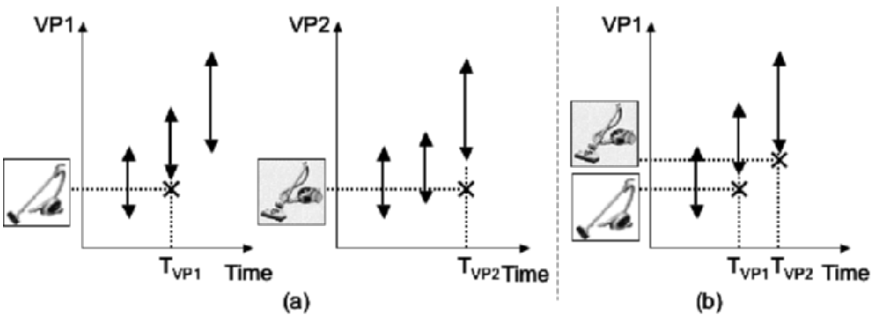


Figure 11.9 Determining which VP to upgrade and establishing upgrade timing

upgrade before TVP1, the first displacement point. In addition, designers must ensure that the timing of the second upgrade agrees with the adopted design solution. The displacement of two design solutions is shown in Figure 11.9(b). The displacement point of each solution is different, which influences the entire upgrade schedule.

Determining the basic specification. After the VP to be upgraded and the upgrade timing have been determined, the designers search for the key components that are to be replaced. By changing these components, they can determine the basic specifications of the target product.

Searching for a design solution. In addition to the key components, designers also have to select the other components to be replaced. These are not the key components, but their replacement is indispensable when searching for design solutions. Designers explore the component database to identify these non-key components and define a design solution for the second generation that would satisfy the basic specifications given above. Designers can reject every design solution that is below the lower limits of the VP ranges. If they can obtain appropriate solutions, they search for solutions for the next generation. If they cannot, they correct the basic specification and design solution for the former generation and try again to find solutions (Shimomura et al. 1999). This operation is based on the methodology proposed by Umemori (Ishigami et al. 2002; Umemori et al. 2001). The designers should continue until every user-demand pattern is satisfied during the upgrade span.

Evaluating an upgrade plan

In this step, the designers make upgrade product lineups composed of design solutions that they have obtained in the previous steps. The product lineup information includes a list of upgraded VPs, the schedule for the upgrades and a list of replacement components.

If there are multiple upgrade product lineups for one user-demand pattern, designers evaluate them on the basis of their cost throughout the upgrade plan, as well as the components-sharing rate, that is, the rate of the components that are identical between generations, and their flexibility to meet future disturbances. They then choose one lineup for one user-demand pattern. The process of upgrade planning ends with the complete upgrade product lineups for each user-demand pattern.

11.5 Alterations to the upgrade plan

After the production or distribution of an upgradeable product, unexpected events may happen. Such events are called ‘disturbances’ in the present study. One such disturbance is fluctuations in future technological trends. We describe a disturbance as a change in the component database. On the basis of this change, designers alter the user-demand patterns and the upgrade plan in accordance with their assumptions about user demands. This section describes the procedure for upgrade plan alterations using two examples of disturbances: *case 1,* production stoppage of components, and *case 2,* emergence of new technology.

Figure 11.10 summarises the processes that take place when an upgrade plan is altered in these two cases.

(1) Description of disturbance. In this step, the designers adapt the component database to reflect the disturbance. In case 1, the designers

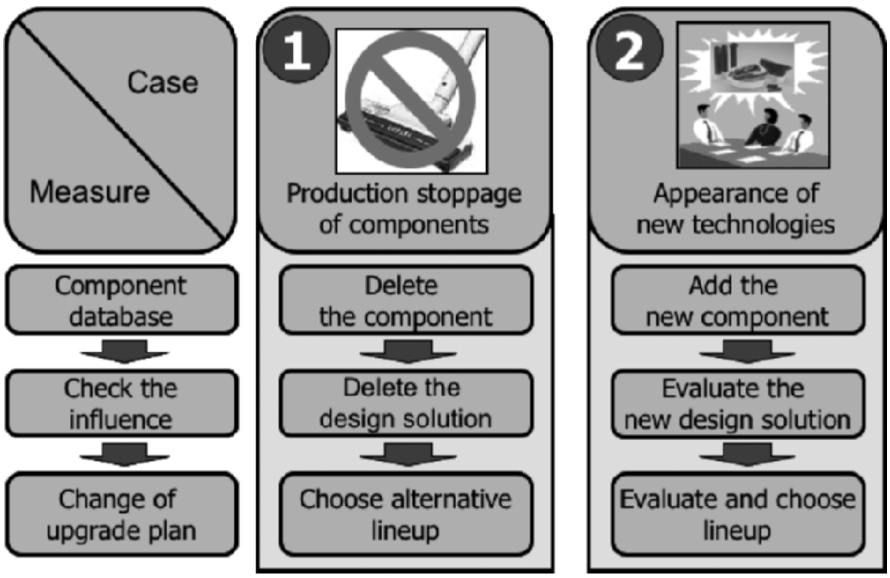


Figure 11.10 Alteration of upgrade plan

delete the pertinent components from the component database. In case 2, the designers add a new component with new technology to the database.

(2) Determining the influence of the disturbance. In this step, the designers determine the influence of a disturbance on every existing upgrade plan for each user-demand pattern. In case 1, the designers check whether any design solution with the pertinent component exists in the design solution lineup. In case 2, the designers determine whether the new design solution with the new component is a good replacement of the existing design solutions in the lineup. This judgment involves evaluating not just a single solution but the entire upgrade design solution lineup.

(3) Altering the upgrade plan. Finally, the designers alter the upgrade plan on the basis of the results of their assessment in the previous step. The alteration occurs in response to a disturbance caused by structural changes. In case 1, designers discard the related design solution, by correcting the upgrade plan and searching for alternative design solutions again. The alternative design solution must of course satisfy the required VP ranges for each user-demand pattern.

Furthermore, the solution must be consistent with the entire upgrade plan. In case 2, the designers replace an existing design solution with a new one only if that replacement improves the upgrade plan in terms of the evaluation described in 11.4: Evaluating an upgrade plan, i.e., in terms of the cost and the components-sharing rate throughout the upgrade plan.

11.6 Applying the upgrade design – a case study

This section presents and explains an example of the upgrade design based on the above-mentioned methodology.

Design conditions

The initial design conditions for upgrade design are summarised in Figure 11.11 and Tables 11.3, 11.4 and 11.5. The details of this condition are discussed below.

Structure model. The example of upgrade design used here is a simple vacuum cleaner model. Its structure model is shown in Figure 11.11. The model uses four VPs: 'Cleanliness', a positive parameter which stands for clean emissions; 'Noise', a negative parameter which stands for the noise

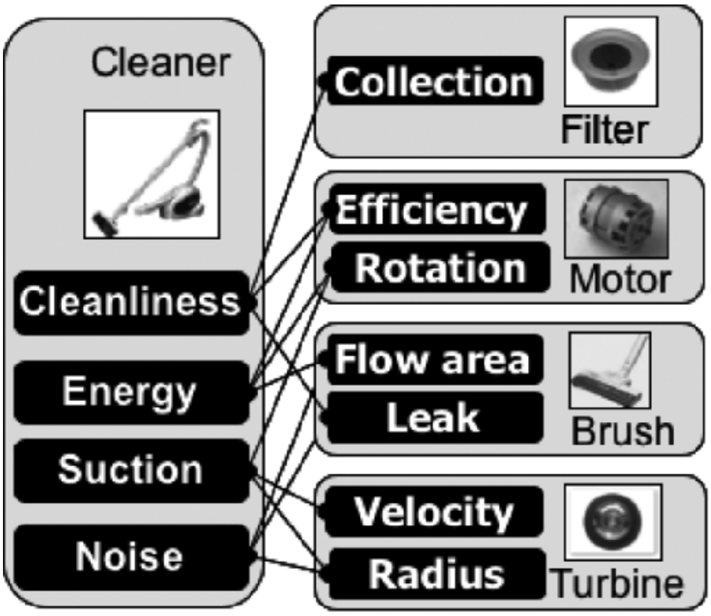


Figure 11.11 Structure model of a vacuum cleaner

generated when the vacuum cleaner is used; 'Suction power', a positive parameter which stands for the vacuuming power of the appliance; and 'Energy consumption', a negative parameter which refers to the power consumed by the vacuum cleaner.

Component database. Tables 11.3 and 11.4 show the design information for a 'Filter' and a 'Motor' in the structure model of the vacuum cleaner presented here. The filter has one DP, called the 'Collection rate', and the motor has two DPs, 'Rotation' and 'Efficiency'.

User-demand pattern. Table 11.5 presents three qualitative user-demand patterns: (1) Commercial users. This includes people who use high-performance vacuum cleaners, and require vacuum cleaners with high suction power and very clean emissions. The other parameters, such as noise and energy consumption, are unimportant. (2) Energy-saving users. This type of user requires machines that consume little energy. Such users demand low energy consumption and do not consider the other VPs. (3) Intermediate users. This pattern represents the middle-level user, who is interested in an upgrade. They require a middle-level performance for all VPs.

Table 11.3 Design parameters of a filter

Filter		F1	F2	F3	F4
DP	Collection rate	0.3	0.5	0.8	0.9
Existence span		[1-2]	[1-3]	[2-3]	[3-3]

Table 11.4 Design parameters of a motor

Motor		M1	M2	M3	M4	M5	M6
DP	Efficiency	0.5	0.7	0.5	0.7	0.5	0.7
	Rotation	120	130	150	150	180	180
Existence span		[1-2]	[2-3]	[1-3]	[2-3]	[2-3]	[3-3]

Table 11.5 User demand patterns for a vacuum cleaner

	Energy	Suction	Noise	Cleanliness
Commercial	H	H	H	H
Energy saving	L	L	H	L
Intermediate	M	M	M	M

Upgrade planning

Under the initial design conditions discussed in the section ‘Design Conditions’, an upgrade plan for a vacuum cleaner was developed, with an upgrade span of three years. Figure 11.12 depicts this upgrade plan, including the VP roadmap and the design solution lineups for each user-demand pattern in the roadmap view. Figure 11.12 (a) relates to suction power, while (b) is about energy consumption. Three narrow bands for each generation show the user demand ranges of the VP. The circles in the bands represent the values of upgrade design solutions for each. The large rectangles behind the user demand range bands show the VP ranges of all products in the market. Since the suction power is a positive VP, its value range gradually rises. On the other hand, the value range of the ‘Energy consumption’ VP declines annually.

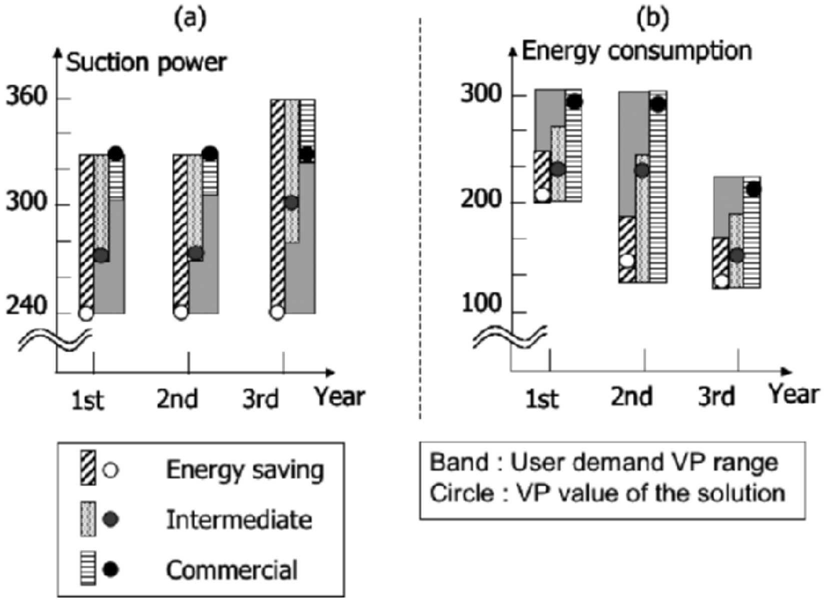


Figure 11.12 Upgrade plan for each user-demand pattern

As for suction power, ‘Energy-saving’ and ‘Commercial’ users do not require any upgrades during the upgrade span. This is because the suction power VP rises gradually, and the VP of a first-generation product is not displaced from the user demand range. Only the intermediate users require an upgrade in the third year.

The VP for energy consumption, the VP decreases annually. The lower limit of the range declines from the first to the second year, so ‘Energy-saving’ users require an upgrade in the second year. In addition, since the upper limit of the value range declines from the second to the third year, ‘Commercial’ and ‘Middle’ users require an upgrade in the third year.

11.7 Discussion

The outcome of the above case study shows that suitable components were selected for each type of customer on the basis of an upgradeable design methodology. Based on these results, product designers can design a product structure that takes future upgrades into account.

The upgradeable design methodology is comparable to some other product design methods. For example, the quality function deployment (QFD) method shows some similarities with the upgrade design methodology, such as the use of a product structure model. However, one of the main differences between the two methods is that the upgrade design methodology includes a process to allow the demands of customers to be predicted on the basis of future technological trends. This provides product designers with greater insight into their customers' preferences.

The upgradeable design methodology can be evaluated by comparing it with some other environmentally conscious business models. The differences between them are shown in Table 11.6.

Table 11.6 Evaluation of environmentally conscious business models

	Environmental impact	**Efficiency**	**Satisfaction with functions**
Upgrade design	Good	Good	Very good
Remanufacturing	Very good	Not good	Not good
Function sales	Good	Good	Good
Material recycle	Not good	Not good	Not good

Conventional environmentally friendly business models, such as remanufacturing and materials recycling, encounter problems with the process of recovering the used products. In addition, recovery operations are expensive. The concept of functional sales, which is a lease service of the products with required functions, is attractive since its function can be replaced based on customer needs.

In comparison to these conventional models, upgrade design can reduce the environmental impact as well as upgrade a product's functionality, without having to deal with the recovery process. The concept of upgrade design is applicable to the functional sales service. Such a concept could result in further reduction of the environmental impact.

The current mass production paradigm contributes significantly to environmental degradation. To solve these environmental problems, concepts involving the reuse and remanufacture of products and materials are being proposed. The design of upgradeable products, which could be used for longer than conventional products and encourage people to reuse artefacts, is one of the most promising approaches using these methodologies. In addition, they might provide new business opportunities in the later stages of their product life cycle. Achieving upgradeable design requires a proper plan, which must include information on the upgradeable design, including when a product should be upgraded, with regard to which function, and to what extent. The upgrade plan should also include a solution lineup for upgradeable design that satisfies these conditions. To devise such an upgrade plan, designers need to predict technological trends and user demands. This paper proposes a methodology for upgrade planning based on the prediction of user demand, and on the assumption that technological trends influence user demands. In addition, a methodology is proposed for changing upgrade plans after target products have been distributed, to meet possible fluctuations in technological trends or user demands.

11.8 Conclusion

This paper proposes a method to design a realistic and feasible upgrade plan which includes predicting future technological trends and user demands by using roadmap information and a component database. A future sudden event (disturbance) is expressed as an alteration to the component database, and a method to effect corrective actions against such disturbances is proposed.

Future studies will focus on the implementation of an upgrade plan alteration system, involving 're-planning' of the upgrade plan based on the updated component database after a future disturbance.

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Cases in Recycling

12 A strategic policy model for promoting secondary materials use

Nur Indrianti, Shinobu Matsuoka and Masaaki Muraki
Department of Industrial Engineering and Management,
Tokyo Institute of Technology, Tokyo, Japan

Abstract

This paper discusses problems associated with the development of a tax and subsidy policy to promote the use of secondary materials converted from industrial waste as a substitute for virgin materials. The main purpose of the study was to examine how a set of tax and subsidy levels modifies consumption and affects welfare. For this purpose, a static partial equilibrium model was developed, taking into account welfare effects in production and consumption under economic and ecological constraints. The results of the study indicate that a virgin materials tax combined with subsidies for waste converters could be an effective policy to promote the use of waste-based secondary materials.

12.1 Introduction

Sustainable development is a concept commonly used to assess the impacts of human activities on nature, the environment and the resource base. The Johannesburg Declaration on Sustainable Development, adopted at the World Summit on Sustainable Development (WSSD) on 4 September 2002 (WCED, 2002), reaffirmed the commitment to sustainable development and urged parties to 'assume a collective responsibility to advance and strengthen the interdependent and mutually reinforcing pillars of sustainable development – economic development, social development and

G. Huppes and M. Ishikawa (eds.), Quantified Eco-Efficiency, 285–305.

environmental protection – at local, national, regional and global levels'. In respect of economic and social development, sustainable production and consumption (SPC) is an essential requirement. SPC is defined as the use of goods and services to meet basic human needs and raise the quality of life while simultaneously minimising the use of natural resources, the use of toxic materials and the emission of wastes and pollutants over a product's life cycle, so as not to risk the ability to meet the needs of future generations (WCED 1987; Hinterberger et al. 1999). In the process of fulfilling human needs and enabling better quality of life, the industrial sector uses virgin materials and energy extracted from natural resources to produce goods. As these resources are normally non-renewable, their depletion is putting our future generations at risk. In addition, both the extraction process and the production activities produce unused by-products or wastes that are harmful to the environment and cause ecological problems. In terms of this problem, Haake et al. (1999) regarded industry as a prime factor in sustainable development.

In response to environmental problems caused by the industrial activities described above, Industrial Ecology (IE) is emerging as an approach aiming at closing the materials cycle to develop industries which minimise the use of resources and the production of wastes (Allenby 1999). This aim is in line with the concept of eco-efficiency, which is to enable more efficient production processes and better products and services while reducing resource use and pollution.

One of the implementations of IE is the development of 'eco-industrial parks' such as that of Kalunborg in Denmark. The principle underlying such eco-industrial parks is to utilise the waste material from one firm as a raw material for another (Graedel and Allenby 1995; O'Rourke 1996; Ehrenfield and Gertler 1997; Lowe and Evans 1995). By utilising waste as a substitute for virgin material, this strategy simultaneously reduces environmental damage due to waste materials and extraction processes, and reduces the rate of exhaustion of natural resources.

In general, existing eco-industrial parks provide symbiotic industrial facilities which allow a chain of waste production and utilisation to be physically set up. However, such arrangements are rare, as most existing firms are not located in close proximity to each other and many types of waste cannot be used directly as raw materials. Consequently, conversion processes are required.

Focusing on raising the productivity of natural resources, Nemerow (1995) and Ayres and Ayres (1996) investigated the possibilities of using several waste materials and found that technical difficulties and economic feasibility can present critical problems, because such sources are often more difficult or more costly to exploit than natural sources. In addition, distant locations increase transport costs and thus make waste materials less competitive. The most promising strategy to implement industrial ecology in this case would be through policy levers.

As regards environmental policy, theoretical and empirical studies have suggested that market-based mechanisms are still the most efficient method to pursue many environmental goals (Ekins and Speck 2000). Considering the interrelationships between the economy, society and the environment, this paper intends to examine the effectiveness of market-based mechanisms in the context of industrial ecology, in particular to promote the use of waste-based secondary materials.

The rest of this paper is organised as follows. The next section reviews the current literature on the subject of market-based environmental policy. This is then followed by a section that presents a model as well as a procedure to determine the optimal level of the policy being considered. Section 4 draws some conclusions from the results of the study, while the final section outlines suggested future research on the topic.

12.2 Market-based environmental policies

The economic literature provides extensive discussions of market-based mechanisms. In discussing the strengths and weaknesses of market-based mechanisms, Bailey (2002) concluded that by allowing flexibility of response, market-based mechanisms generally promote cost-effective environmental protection. Market-based mechanisms, such as environmental taxes and tradable permits, can also be iterated more easily than command-and-control instruments to achieve environmental objectives.

With regard to materials utilisation, Bruvoll (1998) argued that the overuse of virgin materials was induced by their current market prices. Such overuse naturally leads to increased negative environmental externalities and wastes. A tax on the use of materials would help correct these market inefficiencies and would encourage reduced use of virgin materials and hence reduced generation of waste. Such a tax would also increase the prices of products, reduce the demand for and production of products based on those virgin materials and thus stimulate source reduction (Brison 1993). In addition to taxing the use of virgin materials, Pearce and Turner (1993) pointed out that product charges could also be imposed to correct for market failures. Söderbaum (1999) agreed that environmental charges and the tax system certainly represent a powerful tool to move the economy in a more environmentally friendly direction. However, the potential inflationary impact of environmental taxes, such as materials tax or product charges, brought about by the increase in either materials or product prices, may hamper economic performance and competitiveness, which may also reduce social welfare. In response to the high cost of new technologies that facilitate higher recycling rates or less use of natural resources, policy-makers usually consider economic incentives such as direct subsidy. The rationale for such instruments is that they create incentives to achieve goals that are welfare-enhancing (Hahn 2000). Pricing subsidy has been an alternative policy for situations where a government wishes to accelerate the diffusion of new products (Kalish and Lilien 1983; Zaccour 1996; Jørgensen and Zaccour 1999) and to reduce the level of automobile emissions by encouraging the production and pervasive use of bio-fuels (Bard et al. 2000). However, although subsidies have the theoretical potential to achieve environmental objectives at the lowest cost, they require a budget, which will increase public expenditures. Such increases may be at the expense of provisions for social welfare.

Regarding the potential of shifting environmental taxes, particular attention has been paid to shifting taxes from societal ‘goods’, such as income and employment, as well as welfare, to environmental ‘bads’, like pollution and resource degradation. It has been argued that the combination of environmental tax and using the increased revenue to alleviate other taxes would result in a ‘double dividend’ of less environmental damage and a stronger economy (Marshall 2000). Several types of tax shifting have been applied to minimise potential negative effects of environmental tax on economic performance and competitiveness, as discussed above, due to inflationary impact. These include recycling revenues from resource tax to reduce payroll tax (Bruvoll 1998), recycling energy tax to promote technology or production innovation, and recycling energy tax to subsidise renewable energy resource development and to support the improvement in energy efficiency (Ekins and Speck 2000).

A number of studies have examined the extent to which a double dividend can be achieved by focusing on the combination of tax and subsidy. Using the concept of internalising externalities, Dinan (1993) analysed the effectiveness of a combined tax and subsidy policy to reduce the amount of municipal solid waste. In this case, the revenue from the tax imposed on producers of goods that may ultimately be disposed of (e.g. newspapers) was used as subsidies, made available to end users of recycled materials. Using an analytical approach, Dinan concluded that a combined disposal tax and reuse subsidy policy was efficient in reducing waste. However, his model insufficiently covered consumers' decisions on the consumption side, compared to that on the disposal side. In addition, the maximum social welfare was represented by the benefits gained by producers of goods that may use recovered materials under optimal production levels and input choices, without considering the social welfare beyond them.

With respect to the effects of recycling principles on the achievement of policy objectives, Jacobsen (2000) assessed the differences between recycling revenue from an economy-wide reduction of corporate tax rates and recycling tax revenue paid by the energy supply sector while subsidising the same sector for its use of a specific CO_2 low-intensity or neutral fuel such as biomass. This study relates to the Danish initiatives to reduce CO_2 emissions, in which the energy supply sector was the main source of emissions. The study showed that recycling CO_2 tax revenue as a subsidy for biomass use has a much greater reduction effect than the other recycling mode. The result of the study showed that although the policy is considered capable of reducing the level of emissions, both revenue recycling modes create a loss of gross domestic product (GDP) due to the loss of overall efficiency associated with the changed fuel mix in industry and the loss of international competitiveness.

As regards welfare indicators, it has been recognised that Gross Domestic Product (GDP) does not correlate well with changes in national well-being. One of the shortcomings of GDP is its failure to account for activities that occur outside the market, such as the value of natural capital or the harmful effects of pollution on human health. (Hamilton 1999). To counter this shortcoming, the Index of Sustainable Economic Welfare (ISEW) and the Genuine Progress Indicator adjust the GDP to account for factors that contribute to environmental degradation, in particular the depletion of non-renewable resources and the consequent long-term environmental damage.

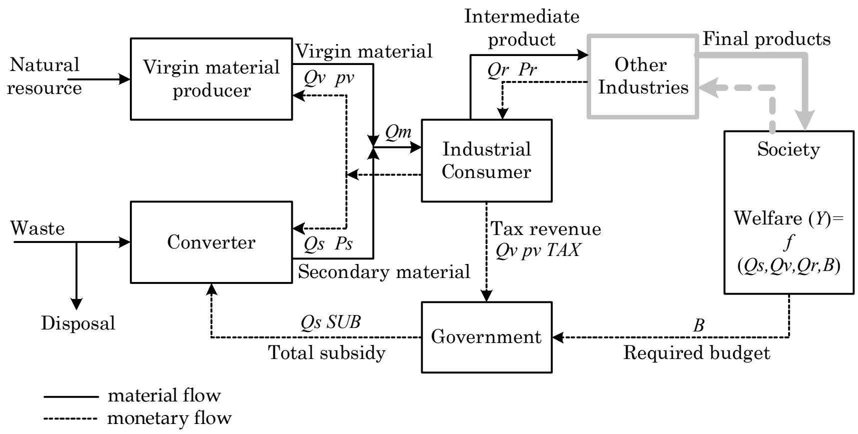

Figure 12.1 Conceptual model of tax and subsidy policy for promoting waste-based secondary material

With regard to the environmentally relevant welfare indicator described above, this paper aims to develop an effective market-based policy for promoting secondary materials, taking both economic and environmental goals into account. We define sustainable economic welfare as the goal of the policy, instead of focusing on only one aspect as the objective of the policy. In particular, the policy is intended to improve environmental performance without damaging the economy.

12.3 The model

Model description

As mentioned above, several environmental taxes, such as resource tax or disposal tax, have been recognised as effective instruments to reduce environmental impacts associated with virgin material use. With regard to the concept of double dividend, this paper focuses on resource tax imposed on virgin materials, using the revenue from the tax to subsidise secondary materials producers or converters. This revenue recycling would reduce the demand on the public budget used to promote the use of secondary materials. The surplus revenue is used to reduce socially undesirable options such as personal income tax. This should provide the policy with the potential to improve welfare.

A strategic policy model was developed to examine the effectiveness of the proposed policy scheme. Addressing the limitations of the earlier tax and subsidy policy models described in the previous section, our strategic policy model encompasses the behaviour of producers and consumers, while taking welfare effects in production and consumption into consideration. As can be seen from Figure 12.1, the model deals with the case in which converters produce waste-based secondary materials as substitutes for virgin materials.

The composite materials are then used by industrial consumers to produce intermediate products, which are then used by other industries to produce final products for public consumption. An example of this case is the recovery of fluosilicic acid from phosphate rock to substitute acid grade fluorspar used to manufacture hydrofluoric acid, from which most fluorine chemicals are derived. It is assumed that the production of the intermediate product accounts for the greatest use of the virgin material, and the implementation of tax is limited to its use for producing the product. Thus, the policy does not affect prices of other sectors. It is also assumed that domestic demand for virgin materials is currently satisfied fully by domestic producers and that neither imported nor exported virgin materials or intermediate products are involved.

As mentioned above, the objective of the model is to maximise sustainable economic welfare, which constitutes the achievement of economic and environmental performance. In this study, we use the term socio-economic welfare, represented by per capita personal income, to indicate economic performance. In selecting the per capita personal income as a welfare-relevant factor, we refer to the Atkinson-style utility function that transforms a nation's income level into a measure of social welfare (England 1998). In addition, per capita personal income is often used as an indicator of the character of consumer markets.

Furthermore, we assume that the virgin materials are extracted from minerals having a relatively short reserve life, so that it makes a significant contribution to the welfare. It is also assumed that a standard has been imposed on the disposal of waste raw materials. In addition, we assume that the properties of a secondary material are the same as those of the virgin material, and that the environmental impacts of the disposal process of waste raw materials are larger than those of the conversion process.

Thus, environmental performance measured in this study by the use of virgin materials, and is measured by the depletion costs and environmental costs associated with the extraction process of such materials. It has been recognised that the process of mining contributes considerably to environmental impacts. The environmental costs considered in this study refer to the following aspects: land disturbance, including soil erosion, and hydrological and landscape changes in the area; dust pollution; and impact on biodiversity (vegetation and habitat destruction, soil and water contamination, etc.). The model is therefore represented by the following:

$$\max \left\{ Fg(TAX, SUB) = \phi Y^{\theta_y} Qv^{\theta_v} \right\} \tag{12.1}$$

Subject to:

$$Ps = fps + cs - SUB \tag{12.2}$$

$$B = SUB\, Qs - pv\, TAX\, Qv \tag{12.3}$$

$$Y = yo\left(1 - \frac{1}{\gamma}\frac{B - bo}{bo}\right) + \tau_s\, Qs + \tau_v\, Qv + \tau_r\, Qr \tag{12.4}$$

$$Qr\, Pr = \beta_o\, Pr + \beta_1\, Y \tag{12.5}$$

$$Pr\, Qr = Qs\, Ps + Qv\, pv(1 + TAX) + Qr\, cr \tag{12.6}$$

$$\ln Qs = \alpha_o + \ln Qm + \alpha_1 \ln\left(\frac{pv(1 + TAX)}{Ps}\right) \tag{12.7}$$

$$Qr = Qm / \mu_r\,,\ Qm = Qs + Qv \tag{12.8}$$

$$Ps \le pv(1 + TAX) \le poc \tag{12.9}$$

$$Y \ge yoo\,,\ Fg \ge fgo \tag{12.10}$$

$$Qv \le vrg \tag{12.11}$$

$$Qs \le qsm \tag{12.12}$$

$$TAX, SUB, Ps, Pr, Qs, Qv \ge 0 \tag{12.13}$$

As shown in Equation (12.1), the efficiency gain of the policy is a function of socio-economic welfare and the use of virgin materials. Equation (12.2) describes the converter's pricing strategy. It shows that the market price of a secondary material depends on its conversion cost, the expected profit per unit of secondary material and subsidies received from the government.

Equation (12.3) shows that the budget required to implement the policy depends on tax revenues and the total subsidy paid to the converter. In this case, it is assumed that under perfect information conditions, the policy-maker can predict virgin material prices.

Approximating the per capita personal income by a linear function and assuming constant values for other factors such as incomes from other sectors, Equation (12.4) shows that per capita personal income is affected by two elements: government expenditures to subsidise converters and the income from production activities. The first term of the equation describes that government's spending to implement the policy affects the per capita personal income. This is because personal income and payroll taxes would be the main sources of the budget. In this case, we chose the concept of marginal excess burden (MEB) to describe the relationship between the budget and the per capita personal income. Here we introduce the current per capita personal income value (excluding the income from the production of virgin and secondary materials, and the production of intermediate products) and the current government's spending as benchmarks for the per capita personal income and for government spending, respectively.

The second, third and fourth terms of Equation (12.4) represent the contribution of the production of a secondary material, of the virgin material, and of the intermediate product to the per capita personal income associated with wage and salary or other incomes from labour. Production activities in the secondary materials sector contribute to the personal income not only through wage and salary, but also through the reduction of disposal costs. This production effect implies the need for caution in determining the level of tax. High tax levels lead to considerable reductions in the use of virgin materials, and hence to improved environmental performance. On the other hand, reduced use of virgin materials means reduced production activities in the virgin materials sector, which may reduce socio-economic welfare.

In maximising utility, Equation (12.5) shows a linear expenditure system (LES) representing the demand for the intermediate product. In addition, Equation (12.6) shows that the production function of the industrial consumer is made up of material costs and production costs. It describes the inflationary impact of the price of intermediate products on the consumption of virgin and secondary materials, which contribute to socio-economic welfare (Eq.12.4). Based on the assumption that the tax and subsidy policy does not affect prices in other sectors, as noted above, this study does not consider the inflationary impact of other products beyond the sectors being considered.

In the case of materials substitution, there is no doubt that relative price changes play an important role in the substitution processes. In the model, we employ a substitution equation based on the demand function used by Spilimbergo and Vamvakidis (2003), and assume that materials are separable from other inputs such as capital, labour and energy. Equation (12.7) shows that the demand for secondary materials depends on the total demand for the composite materials, and material substitution is assumed to be taking place when the relative price of a secondary material is more favourable than the after-tax price of the corresponding virgin material. As shown in Equations (12.5) and (12.6), inflationary impact due to virgin materials tax increases the price of intermediate products and thus reduces the demand for them. As a result, the demand for the composite material will decrease, in turn reducing the demand for the secondary material. Consequently, production activities will decrease, resulting in reduced welfare as described by Equation (12.4).

Equation (12.8) describes the materials balance in relation to the composite materials used by industrial consumers. Equation (12.9) shows that the substitution processes require a condition in which the price of a secondary material is at least the same as the after-tax price of the virgin material. With respect to domestic commodity competitiveness, this study considers the external competitors' price (including import tariff) as the maximum price level at which the virgin material is sold.

Strategic constraints (12.10) imply limiting conditions of socio-economic welfare and sustainable economic welfare to be achieved after the policy is implemented. It stresses that improved environmental performance does not necessarily reduce economic performance.

Equation (12.11) describes the upper limit of virgin materials use, which is regarded as an ecological constraint on the policy target. This implies rising welfare at reduced resource use. Inequality constraint (12.12) imposes an upper limit on the secondary materials production, based on waste availability as the raw materials input. It is assumed that price and demand for the secondary material do not affect the price and production level of the main product, in which waste is produced as a by-product. Finally, we assign a set of non-negative constraints in Equation (12.13).

Numerical illustration and solution procedure

The model is a nonlinear program. In view of its structure, including the logarithmic demand function (Equation (12.7)), it is most likely that analytical methods will fail to find an optimal solution. Therefore, we resort to a numerical method, based on the mathematical solution provided in the Appendix to this paper.

To examine how tax and subsidy policies modify consumption and affect welfare, the model was simulated using a set of data presented in the Appendix. As concerns the values of the data, verifications were applied to ensure that the values of the parameters and variables can be varied without changing the conclusion. The simulation resulted in the following initial condition: qvo=141.53x10^6 kg, qro=78.63x10^6 kg, yoo=USD 1242.20, and fgo=USD 1042.93.

We then simulated the model to examine the effectiveness of a stand-alone subsidy or tax policy. When applying a subsidy policy (zero tax), USD1.5 kg^{-1} of subsidy is required to satisfy the constraint of substitution process, in which the maximum selling price of a secondary material equals the price of the virgin material (Eq. 12.9). The subsidy results in Fg=603.433 and Y=464.51, both less than in the initial condition. When applying a tax policy (zero subsidy), 33.4% tax is required to satisfy the substitution process requirement. However, this tax level does not satisfy the constraint of maximum allowable tax (16%), in which the after-tax price of the virgin material must be the same as the external competitor's price. These results show that a tax or subsidy policy on its own would not be an effective option for the case being considered.

To examine the effectiveness of a combined tax and subsidy policy, the model was then simulated based on various levels of tax and subsidy. The results of the simulation are depicted in Figures 12.2 – 12.5. Figure 12.2 shows that sustainable economic welfare increases with tax and decreases with subsidy.

While high tax leads to great reductions in virgin materials use, small subsidies lead to small budgets. Therefore, as can be seen from Figure 12.3, socio-economic welfare also increases with tax and decreases with

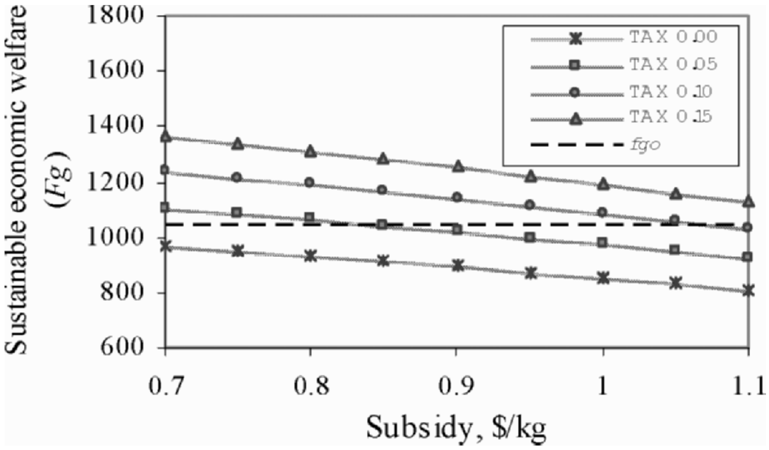

Figure 12.2 The effect of tax and subsidy policy on sustainable economic

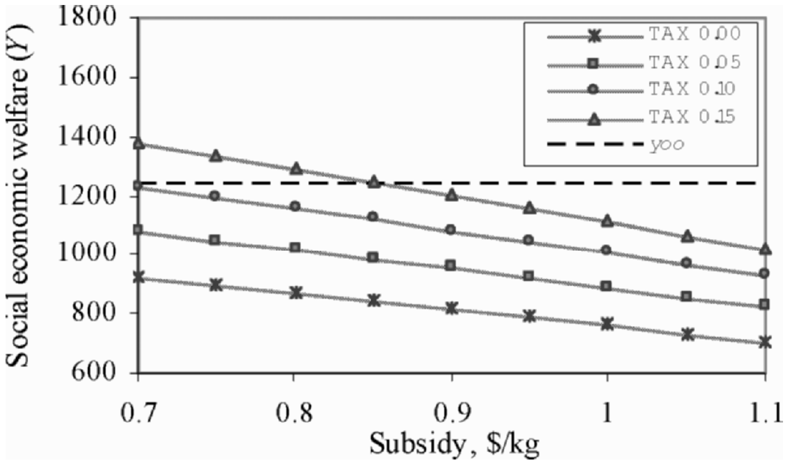

Figure 12.3 The effect of tax and subsidy policy on social economic welfare

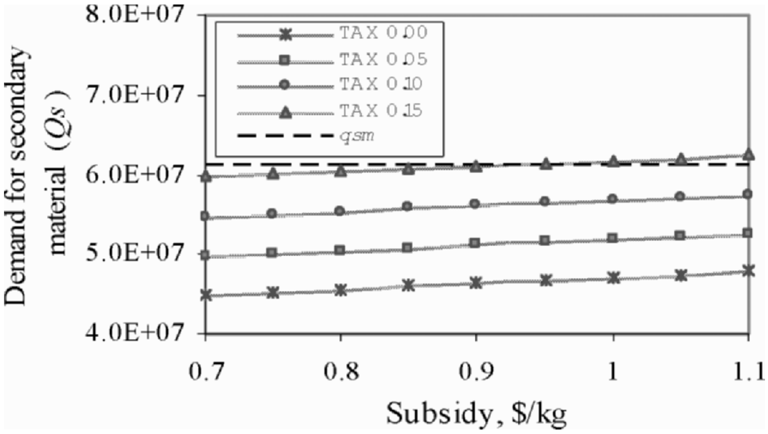

Figure 12.4 The effect of tax and subsidy policy on the demand for secondary material

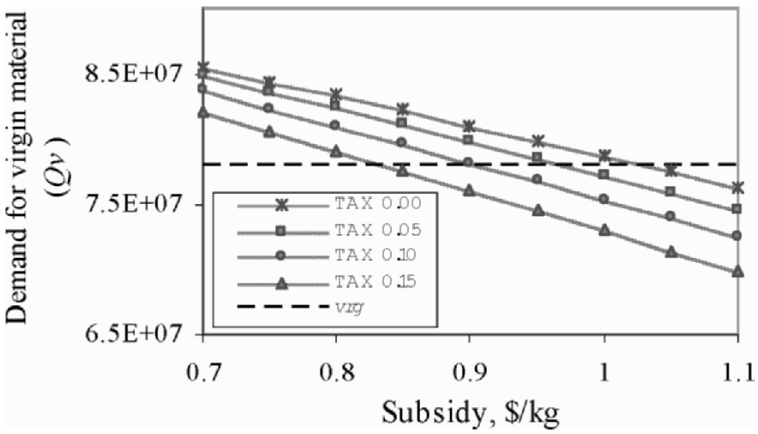

Figure 12.5 The effect of tax and subsidy policy on demand for virgin material

welfare subsidy. Figure 12.4 shows that the demand for secondary materials increases with tax and subsidy, while Figure 12.5 shows that the demand for virgin materials decreases with tax and subsidy. However, Equation (12.2) indicates that a small subsidy will make the secondary material less competitive and lead to a small demand for it (Equation (12.7)). Moreover, Eq. (12.9) implies that there are upper limits on tax and lower limits on subsidy as a requirement of the substitution process. Recognising these, we developed a solution procedure to determine the optimal tax and subsidy levels. The basic steps of the solution procedure are presented below.

1. Begin by calculating *fgo*, *yoo*, and the maximum allowable tax (*txm*). The maximum allowable tax is defined as the tax rate at which the price of the virgin materials equals the external competitor's price.

$$pv(1+TAX) = pco \tag{12.14}$$

2. Calculate the minimum subsidy *submin* based on the tax value resulting from step 1. The minimum subsidy is defined as a subsidy rate at which the price of the secondary material equals the price of the virgin material after tax.

$$Ps = pv(1+TAX) \tag{12.15}$$

3. Consider the following condition with a subsidy step size (Δsub), starting from *submin:*

3ia. $Y<yoo$ OR $Fg<fgo$ 3ib. $Y>yoo$ OR $Fg>fgo$

3ii. Qs>qsm AND Qv>vrg

3iii. Y$\geq$yoo AND Fg$\geq$fgo AND Qs$\leq$qsm AND Qv$\leq$vrg

If *TAX* = *txm* and condition (3ia) prevails,
there will not be any feasible solution.

If *SUB* = *submin* and condition (3ia) prevails,
there will not be any feasible solution.

If conditions (3ib) and (3ii) prevail,
reduce TAX by a step size Δtax and repeat step 2.

If condition (3ii) prevails,
the optimal solution has been found.

Using this procedure, the optimal solution for the case being considered was found at:

TAX=15.2% and SUB=0.84

with Fg=1294.11, Y=1261.37, Ps=5.16, $pv(1+TAX)$=5.18,

$Qv=77.8\text{x}10^6$, $Qs=60.9\text{x}10^6$, $Qr= 77.1\text{x}10^6$, $\text{B}=-2.08\text{x}10^6$

The above result indicates that combined tax and subsidy could be an effective policy for the case being considered. It could improve sustainable economic welfare without reducing socio-economic welfare. As can be seen from the model, the conversion cost of secondary materials is one important determinant of whether environmental and economic goals can be achieved simultaneously. The higher the conversion cost, the higher the subsidy required, resulting in high tax levels. If the total subsidy cannot be achieved by applying maximum allowable tax, the public budget will increase and socio-economic welfare will be reduced. Other important determinants of the socio-economic welfare are the share factors of production activities by the intermediate product, virgin material, and secondary material sectors, representing how each sector contributes to per capita personal income.

12.4 Conclusion

This paper has discussed the effectiveness of a combined tax and subsidy policy to promote the use of waste-based secondary materials as substitutes for virgin materials. To examine how specific tax and subsidy levels influence production and consumption and affect welfare, a strategic policy model was developed, taking into account the interconnection between resource consumption and the socio-economic system.

The result of the study indicates that a combined tax and subsidy policy could stimulate the use of waste-based secondary materials without imposing a burden on the economy. An increase in socio-economic welfare and the substitution process induced by the policy would increase industrial eco-efficiency. Increased socio-economic welfare means increased value of industrial output. A reduction in waste and reduced use of virgin materials resulting from the substitution process means reduced depletion and environmental costs and hence reduced input costs, assuming no significant influences on emissions.

In terms of the impacts of the policy, imposing a tax on virgin materials would probably result in job losses in the mining sector. On the other hand, encouraging waste mining through subsidy will create jobs. In addition, the potential employment growth in the waste conversion sector could simultaneously create markets for alternative technologies and in turn create further new employment.

Developing partnerships with suppliers could be another option to implement such a policy more efficiently, especially in terms of administrative cost related to materials flow monitoring systems.

12.5 Future research agenda

This paper has been concerned with a static model of tax and subsidy policy to promote the use of waste-based secondary materials, whereas the concept of sustainability implies long-term planning. Hence, the existence of dynamic factors must be taken into account when developing the policy. In this case, one important dynamic factor could be industrial learning, which encompasses increasing efficiency of labour via improved methods or technology change. Such learning would affect the price of virgin and secondary materials, and thus influence their production and consumption. This would also result in changes in the optimal level of the policy. This dynamic situation necessitates the development of a strategy for implementing the policy on a flexible basis, so that the level of the policy can be adjusted if necessary. Future research will need to focus on this issue.

By assuming that there is no quality difference between virgin and secondary materials, the model presented in this paper assumes that environmental impacts resulting from the use of virgin materials are the same as those resulting from the use of secondary materials. In practice, however, they may result in different environmental impacts, in particular in the production or consumption stages. Future work should incorporate this issue in the model.

Finally, the model discussed in this paper did not consider international trade, either import or export. To incorporate foreign trade into the model, modifications would be required, in particular for the demand function. In this case, the elasticity of substitution of products coming from different countries and the exchange rates should be taken into consideration. This is another subject for further study.

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Appendix

Mathematical solution

Substituting *Ps* (Eq. (12.2)) and *Qm* (Eq. (12.8)) in *Qs* (Eq. (12.7)) yields:

$$Qs = f(TAX, SUB, Qv) = \frac{e^{\alpha_o} Qv \left(\dfrac{pv(1+TAX)}{cs + fps - SUB} \right)^{\alpha_1}}{1 - e^{\alpha_o} \left(\dfrac{pv(1+TAX)}{cs + fps - SUB} \right)^{\alpha_1}} \qquad \text{(i)}$$

Rewriting *Y* (Eq. (12.4) and *Pr* (Eq. (12.5)) using Eqs. (12.2) and (12.8):

$$Y = yo\left(1 - \frac{1}{\gamma}\frac{B - bo}{bo}\right) + \tau_s Qs + \tau_v\, Qv + \tau_r\,(Qs + Qv)/\mu_r \qquad \text{(ii)}$$

$$Pr = \frac{Qs\,(cs + fps - SUB) + Qv\; pv(1+TAX) + cr\,(Qs + Qv)/\mu_r}{(Qv + Qs)/\mu_r} \qquad \text{(iii)}$$

Substituting Eqs. (12.8), (ii) and (iii) in Eq. (12.5) yields:

$$Qv = \frac{bo\mu_r \begin{pmatrix} yo\beta_1\left(1 + \gamma - e^{\alpha_o}\Phi^{\alpha_1}(1+\gamma)\right) + \\ \gamma\beta_o \begin{pmatrix} cr(1 - e^{\alpha_o}\Phi^{\alpha_1}) + pv\mu_r(1+TAX)(1 - 2.2^{-2\alpha_o}\, 10.8731^{\alpha_o}\,\Phi^{\alpha_1} \\ + e^{2\alpha_o}\Phi^{2\alpha_1}) + \mu_r(cs + fps - SUB)(e^{\alpha_o}\Phi^{\alpha_1} - e^{2\alpha_o}\Phi^{2\alpha_1}) \end{pmatrix} \end{pmatrix}}{\begin{pmatrix} bo\gamma \begin{pmatrix} cr + pv\mu_r(1+TAX)(1 - e^{\alpha_o}\Phi^{\alpha_1}) + e^{\alpha_o}\Phi^{\alpha_1}\mu_r(cs + fps - SUB) \\ -\beta_1(\tau_r + \mu_r\tau_v(1 - e^{\alpha_o}\Phi^{\alpha_1}) + e^{\alpha_o}\Phi^{\alpha_1}\mu_r\tau_s) \end{pmatrix} \\ -\, yo\beta_1\mu_r\left(pvTAX(1 - e^{\alpha_o}\Phi^{\alpha_1}) - e^{\alpha_o}\Phi^{\alpha_1} SUB\right) \end{pmatrix}} \qquad \text{(iv)}$$

where e=2.71828 and $\Phi = \dfrac{pv(1+TAX)}{cs + fps - SUB}$.

Using the zero value of TAX and *Qs,* and substituting Eq. (12.8) into Eq. (12.6) yields:

$$pro = cr + \mu_r \; pv \tag{v}$$

Substituting Eqs. (12.4), (12.8), and (v) in Eq. (12.5) yields:

$$qvo = \left(yo(1+\frac{1}{\gamma})\beta_1 + pro\;\beta_o \right) \Big/ \left(\frac{1}{\mu_r}(pro - \beta_1\;\tau_r) - \beta_1 \tau_v \right) \tag{vi}$$

and $\quad qro = qvo \,/\, \mu_r$ (vii)

From Eqs. (vii) and (12.4), we obtain:

$$yoo = yo\,(1 + (1/\gamma)) + qro\;\tau_r \; + qvo\;\tau_v \tag{viii}$$

At current condition, *Qs* is zero, thus *qmo* is equal to *qvo,* and therefore:

$$fgo = \phi \; yoo^{\theta_y} \; qvo^{\theta_v}$$

Data used for simulation

pv	=	4.50	ϕ	=	1787.74	τ_r	=	$1.06\text{x}10^{-7}$
pco	=	5.22	θ_y	=	0.82	τ_v	=	$8.24\text{x}10^{-8}$
cs	=	5.20	θ_v	=	-0.34	τ_s	=	$5.88\text{x}10^{-8}$
fps	=	0.80	β_o	=	$5.68\text{x}10^{7}$	α_o	=	-0.83
cr	=	5.00	β_1	=	$0.23\text{x}10^{6}$	α_1	=	1.45
μ_r	=	1.8	γ	=	4.5	qsm	=	$6.1\text{x}10^{7}$
o	=	$2.20\text{x}10^{7}$	yo	=	1000.00	vrg	=	$7.8\text{x}10^{7}$

Decision variables

TAX	:	tax rate (%)
SUB	:	the amount of subsidy given to a converter per kg of secondary material produced (USD kg^{-1})
Initial conditions		
pro	:	current price of intermediate product (USD)
qvo, qro	:	current use of virgin material, and intermediate product (kg)
foo	:	current value of sustainable economic welfare (USD)
yoo	:	current value of socio-economic welfare (USD)
Endogenous variables		
Fg	:	sustainable economic welfare (USD)
Y	:	socio-economic welfare (USD)
Qv, Qs, Qr	:	quantities of virgin material, secondary material and intermediate product produced (kg)
Qm	:	Total demand for composite materials (kg)
Ps	:	market price of secondary material (USD kg^{-1})
B	:	budget spending for policy implementation (USD)

Decision variables		
Exogenous variables		
yo	:	benchmark level for per capita personal income, USD
bo	:	benchmark level for government spending, USD
pv	:	price of virgin material, USD kg^{-1}
pco	:	external competitor's price, USD kg^{-1}
cs	:	conversion cost, USD kg^{-1}
cr	:	production cost of intermediate product, USD kg^{-1}
qsm	:	available waste material, kg
vrg	:	maximum use of virgin material, kg
fps	:	expected profit per unit of secondary material, USD kg^{-1}
μ_r	:	intermediate product conversion factor
Parameters and constants:		
ϕ	:	calibration parameter
θ_y	:	elasticity of sustainable economic welfare with respect to socio-economic welfare
θ_v	:	elasticity of sustainable economic welfare with respect to the use of virgin material
β_o	:	minimum consumption
β_1	:	private marginal cost factor
γ	:	marginal excess burden
τ_v , τ_r	:	share factor associated with wage and salary or other incomes for labour from the production of virgin material and intermediate product, respectively.
τ_s	:	share factor associated with wage and salary or other incomes for labour, and associated with other incomes caused by reduction in disposal costs, from the production of secondary material.
α_o	:	a constant term
α_1	:	cross-price elasticity of substitution between virgin and secondary materials

13 Eco-efficiency analysis of the plastic recovery systems in Hyogo eco-town project

Helmut Yabar and Tohru Morioka
Division of Sustainable Energy and Environmental Engineering, Graduate School of Engineering, Osaka University, Japan

Abstract

Japan started the promotion and development of eco-towns in 1997, with the aim of reducing the environmental pressure through a symbiosis of industries and cities. Hyogo prefecture (located in the west of Japan) has been promoting a recycling-oriented society with the cooperation of industries, citizens and businesses.

This study analysed the possibilities of implementing a reverse logistics network for plastics recovery and its coupling with existing industries and technologies. Since many industries are operating in Hyogo, the use of their facilities and/or equipment for plastics recovery has been proposed, specially the steel and chemical industries. Reverse logistics was used to analyse the supply of input plastics in terms of quantity and quality. This means clustering small neighbouring cities for synchronised sorted collection (there is a difference between small cities, which already use sorted collection, and big cities, where no sorted collection for plastics exists), and coordinating activities between larger cities.

This study was part of a research project aimed at closing the loop in the Japanese plastics industry by introducing an integral approach with improvements to the upstream and downstream sides of the plastic supply chain life cycle.

This paper presents the results of the first part of the study, which included the use of domestic plastic packaging as plastic sources and material, feedstock and energy recovery as recovery technologies. The results

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of this study indicate that the application of reverse logistics, combined with the appropriate recovery technologies in Hyogo eco-town, is both environmentally and economically beneficial. However, it requires close collaboration between local governments and the industrial sector.

13.1 Introduction

Fostering sustainable development requires changing linear economies towards a system where production and consumption cycles are closed as much as possible. Strategies for this goal of closing the loops include the re-use and re-integration of products and components, recycling of materials and energy recovery (Indigo Development 2004). Further strategies towards more sustainable production include increasing yields by new or optimised production technologies and using less material to manufacture the same product. All of these options aim to reduce the materials input into a given system (Moriguchi 1999). The reduction of materials requirements, both absolute and relative, is also referred to as dematerialisation.

This paper is part of a research project which aims at closing the loop in the Japanese plastics manufacturing sector. The research project introduces improvement strategies to both the upstream and downstream sides of the sector's supply chain (Morioka et al. 2003).

In this paper, we focus on an eco-efficiency analysis of the plastics end-of-life improvement strategies based on streamlining plastic waste materials and energy recovery. Since the main impediment to improving recycling levels are the collection, sorting and transport costs (Yabar and Morioka 2002), we propose the introduction of reverse logistics as a method to reduce these costs, in order to make plastics recycling an economically attractive option. Coupling the reverse logistics system with an adequate recycling technology, using existing industrial facilities should make it possible to construct a plastic recovery system covering a wide area. The scenarios were driven by the Japanese environmental policy, which set targets to increase recycling levels and reduce final waste disposal. As for the methodology, the environmental and cost impacts of four scenarios were evaluated, using life cycle assessment and life cycle costs, respectively. Both evaluations were normalised and an eco-efficiency analysis was used to identify the best scenario.

Hyogo eco-town

The concept of eco-towns was introduced in Japan in 1997, as a way to promote the symbiosis of industries and cities. Since then, more than 20 eco-towns have been implemented in cities and regions across the country, Kitakyushu being the best known.

Hyogo prefecture has been promoting a recycling-oriented society as part of the restoration efforts after the devastating earthquake of 1994, and has received from many areas and sources. This experience served as a basis to construct a wide area sustainable closed loop system for recycling, involving partnerships of industry, government and citizens. The basic plan of Hyogo eco-town includes (Hyogo Eco Town Promotion Conference 2003a):

- Recycling promotion based on industry cooperation: the steel and the chemical industries (primary industry) are located in the western part of the Hyogo area, while the main consumption area (process assembly type industries) is in the eastern part. The know-how and technologies available at the advanced industries can be used to provide a recycling base for the existing factories.
- Construction of a recycling system covering a wide area in cooperation with other cities and prefectures: the industrial infrastructure and physical distribution facilities available at Hyogo can be used to meet the needs of other areas, in order to promote a recycling system covering a wide area.
- Promotion of citizen cooperation: citizen groups such as the 'environmentally conscious purchasing' group and the 'take your bag for shopping' group have been active in Hyogo for almost a decade. Starting from this basis, the Hyogo eco-town plan also aims at active participation by citizens in the promotion of recycling.

Basic plan of the eco-town and plastics recovery facilities

The main object of Hyogo eco-town is to construct a closed loop recycling system using existing facilities (Hyogo Eco Town Promotion Conference 2003b). This means taking advantage of some key industries that are already operating in the area, such as the steel industry, chemical industry, energy supply industry, etc. However, the focus should not only be on recovery technologies, but also on the logistics of an adequate supply of the input plastics, in terms of both quantity and quality (Yoshinaga et al. 2002). This requires clustering small neighbouring cities for synchronised sorted collection and coordination among larger cities. It is also necessary to determine the best option for the locations and characteristics of Materials Recovery Facilities (MRF) and/or pre-treatment facilities. Finally, the most adequate transport system must be established.

13.2 Reverse logistics approach to plastic recovery

Reverse logistics entails collection, handling and transport facilities to return recyclables to an established recycling centre or a recovery facility (Fleischmann 2001). In order to achieve an efficient reverse logistics network, the following factors should be addressed (De Brito and Dekker 2004):

- Mismatches in demand and supply in terms of timing, amounts and quality of the product.
- Economies of scale must be sufficient to make reverse logistics environmentally and economically viable.
- On the operational side, a cost–benefit analysis must be made of collection, transport, freight, MRF, etc.

Reverse logistics and recovery technologies analysis

In order to establish an efficient reverse logistics network for plastics recovery and then couple it with the existing technologies, the current situation must first be analysed. This means analysing the cities that are already applying sorted collection for plastics, the industries that usually discharge plastic wastes in their processes, the existing technologies for plastics recycling in the area, etc. Subsequently, the main factors affecting the collection system should be determined. These factors include collection areas and frequencies, collection efficiency, transport modes as well as the Materials Recovery Facilities (MRF), which are usually the first destination of the collected waste plastics. The next step is an analysis of the recovery technologies. As mentioned above, there are many industries in Hyogo whose technologies and facilities can be used for the recovery of plastics. The present study focused on the steel and chemical industries. In the case of the steel industry, plastic wastes can replace virgin resources such as coal and coke in the steel-making process. As for the chemical industry, plastics can be transformed back to basic chemical compounds through gasification. After the analysis of all these factors has been completed, possible scenarios for the reverse logistics network and the best plastics recovery technologies can be established. Figure 13.1 shows the reverse logistics flow analysis.

Plastics recovery technologies in Japan

As a result of many years of research and technological development, as well as policy support, Japan now has available many methods and

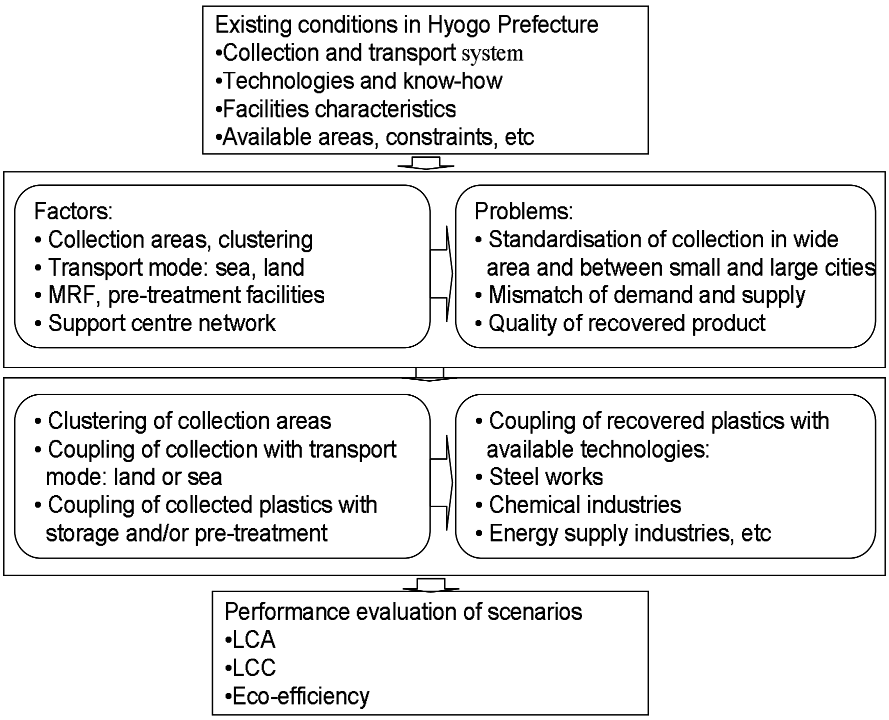

Figure 13.1 Reverse logistics approach to plastic wastes recovery

technologies for plastic waste recycling. These recycling methods, insofar as they are currently recognised under the Japanese plastics containers and packaging recycling legislation (PWMI 2004) include material, chemical and thermal recycling (see Table 13.1). Material recovery technologies include the transformation of waste plastics into flakes for further product making (open loop) and the chemical transformation of polymers into their original monomers (closed loop). In the case of chemical recycling, which is also known as feedstock recycling in Europe (Lundquist et al. 2000); it includes using plastics as a feedstock in blast furnaces, liquefaction, transformation into coke and gasification. Thermal recovery options being considered include incineration with energy recovery, use as fuel in cement industries and transformation into refuse-derived fuel (RDF) pellets. In general, all plastics recovery technologies need some form of sorting and pre-treatment in order to obtain a good quality recovered product (PWMI

Table 13.1 Plastic wastes recycling methods

<table>
<tr><th>CATEGORY</th><th colspan="2">RECYCLING METHOD</th><th>TERM USED IN JAPAN</th></tr>
<tr><td>Mechanical recycling</td><td colspan="2">Plastic raw materials
Plastic products</td><td>Material recycling</td></tr>
<tr><td rowspan="4">Feedstock recycling</td><td colspan="2">Monomerization</td><td rowspan="4">Chemical recycling</td></tr>
<tr><td colspan="2">Blast furnace reducing agent</td></tr>
<tr><td colspan="2">Coke oven chemical feedstock recycling</td></tr>
<tr><td rowspan="2">Gasification
Liquefaction</td><td>Chemical feed-stock</td></tr>
<tr><td rowspan="2">Energy recovery</td><td>Fuel</td><td rowspan="2">Thermal recycling</td></tr>
<tr><td colspan="2">Cement kiln
Waste power generation
Refuse-derived fuel (RDF)</td></tr>
</table>

2002). The government is promoting the chemical recycling of plastics, because of the large treatment capacity available in the steel industry, which already has the know-how and installed facilities, such as coke ovens and blast furnaces.

Recovery technologies evaluation: basket-of-goods method

The basket-of-goods approach (Heyde and Kremer 1999) allows a fair comparison of recovery technologies. In this method, each scenario contains one recovery process and a number of conventional processes (complementary processes) that manufacture the products of recycling and recovery processes included in other scenarios. This way, each scenario produces the full complement of products of all recovery processes in given proportions (basket of goods). This approach allows scenarios to be directly compared in terms of a given indicator, because their waste input and product output are the same (Patel et al. 1999).

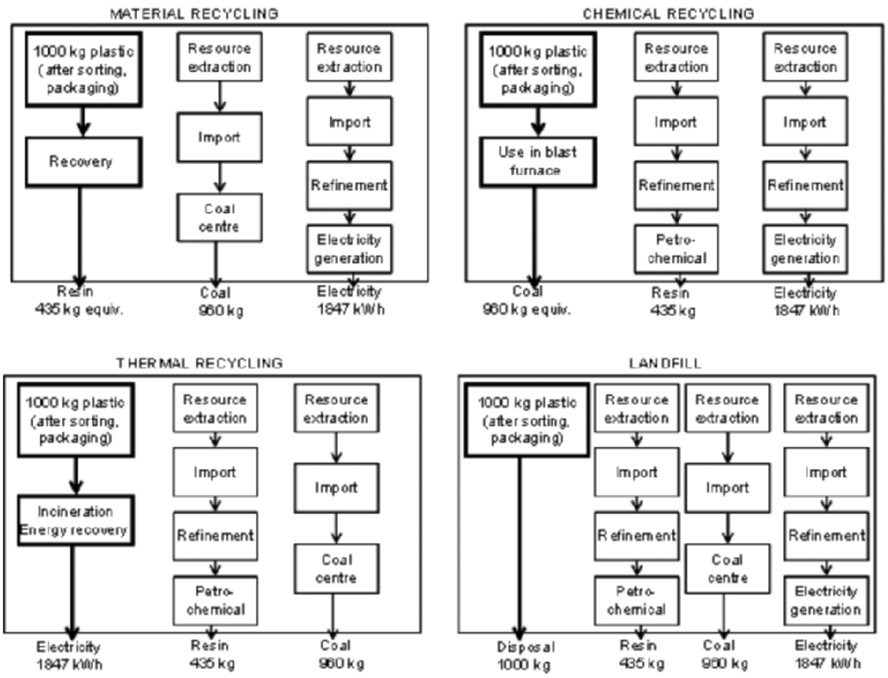

Figure 13.2 Scenarios for plastic recovery technologies

As recovery technologies, our study considered material recycling, chemical (feedstock) recycling and incineration with energy recovery. Figure 13.2 shows the details of the scenarios we analysed. Scenario 1 includes a materials recovery system in which the recyclate substitutes different products made of primary plastics (open loop). It also contains two complementary processes that produce electricity and coal through conventional methods. The same principle applies to the other two scenarios, involving chemical recovery and thermal recovery, respectively. In the case of chemical recovery, we assumed that the plastic wastes would be used as a substitute for coal in a blast furnace, and the thermal recovery scenario assumes energy recovery from a regular municipal solid waste incinerator. Finally, landfill was assumed as a base scenario. The functional unit we used was one ton of plastics (collected, compressed and packed) transported to the recovery facility.

Plastics recovery technologies: environmental impacts and costs analysis

The evaluation of the recovery technologies considered both environmental and economic performance. Indicators used for the environmental performance of the scenarios were virgin resources and energy use, global warming potential and final disposal waste potential (Figure 13.3). The selection of indicators was mainly driven by Japanese environmental law, which sets targets to increase materials recycling by 40% and reduce final waste disposal by 50% by the year 2010 (Yabar and Morioka 2003). Since Japan has committed itself to reducing greenhouse gas emissions by 7% over the period 2008–2012 period, we also included energy use and global warming potential as indicators. The economic evaluation included the costs of recovery technology as well as those of the complementary processes (Figure 13.4). In the case of the materials recovery scenario, 44% recovery efficiency was assumed.

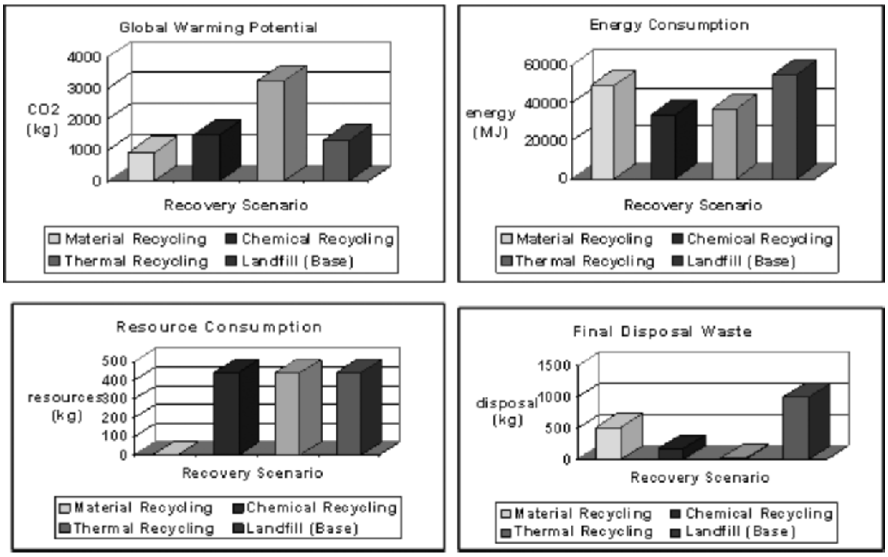

Figure 13.3 Environmental impact analysis of the scenarios

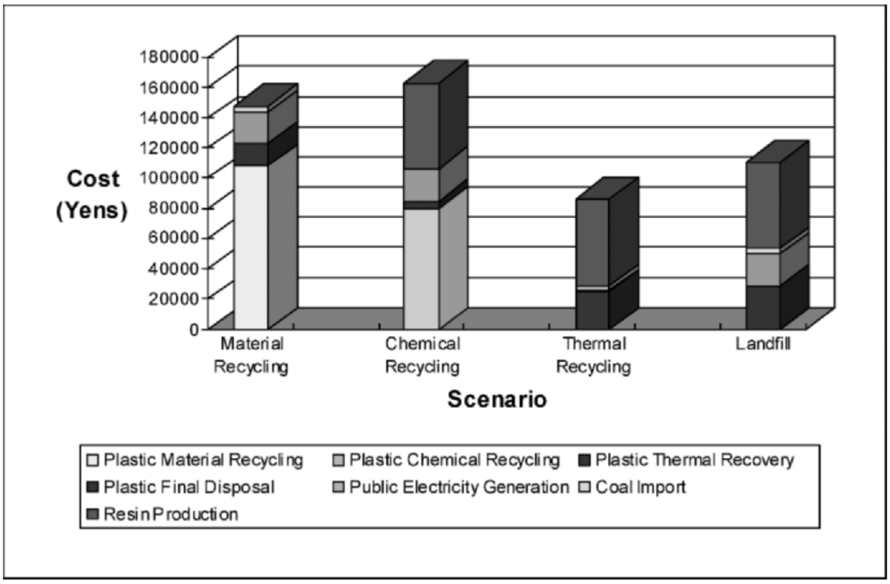

Figure 13.4 Economic analysis of the scenarios

Evaluation of plastics recovery technologies

Although the landfill scenario had the worst environmental performance for three of the four indicators, it is very difficult to estimate from the environmental analysis which scenarios deliver the best and worst environmental performance overall. The economic analysis showed that the material recycling scenario and the chemical recycling scenario are the most expensive ones. One possible reason for the high costs of the material recycling scenario is that we only assumed 44% materials recovery efficiency. Increasing this factor significantly improves the scenario's economic performance. The thermal recovery scenario has the best economic performance, and its environmental impact is comparatively low.

This analysis of the environmental and costs impacts of various recovery technologies provides only a partial view of the plastic waste management problem. In order to propose viable alternative scenarios, it is necessary to analyse the plastic waste management system from a cradle-to-grave perspective, i.e. from the time plastic wastes are collected until they are reclaimed and/or disposed.

13.3 Plastic recovery systems: reverse supply chain analysis

Reverse supply chain analysis takes into account all the steps in the waste plastics flow: from the time the plastic wastes are discarded until their final recovery or disposal. The present analysis focused on the waste collection, sorting and transport phase and its linkage with the most appropriate recovery technologies in order to construct an effective plastics recovery system for Hyogo.

Collection analysis: grid city model

Most collection analyses are based on assumptions of transport distances without considering important factors such as population densities, different collection systems for urban and rural areas, collection types, collection frequencies, truck capacities, distance to transfer stations, etc. The grid city model was conceived to allow a reliable and relatively easy calculation of the environmental and economic impacts of collection (Ishikawa 1996). The basic characteristics of this model are as follows:

Calculation Model

For the purpose of the study, it is assumed that the urban area has a regular grid type configuration. A collection truck starts its journey from the Materials Recovery Facility (which in Japan is called a recycling centre) to its collection area. In the collection area, the truck starts collecting waste from the stations, usually located at every block, until it is loaded to capacity. The collection truck then returns to the transfer station, unloads the wastes and goes back to the collection area to repeat the operation. This means that the collection trip can be expressed in terms of the distance travelled inside the collection area and the round trip to the transfer station. The calculation of the distance travelled by one truck during a round trip is showed in Figure 13.5.

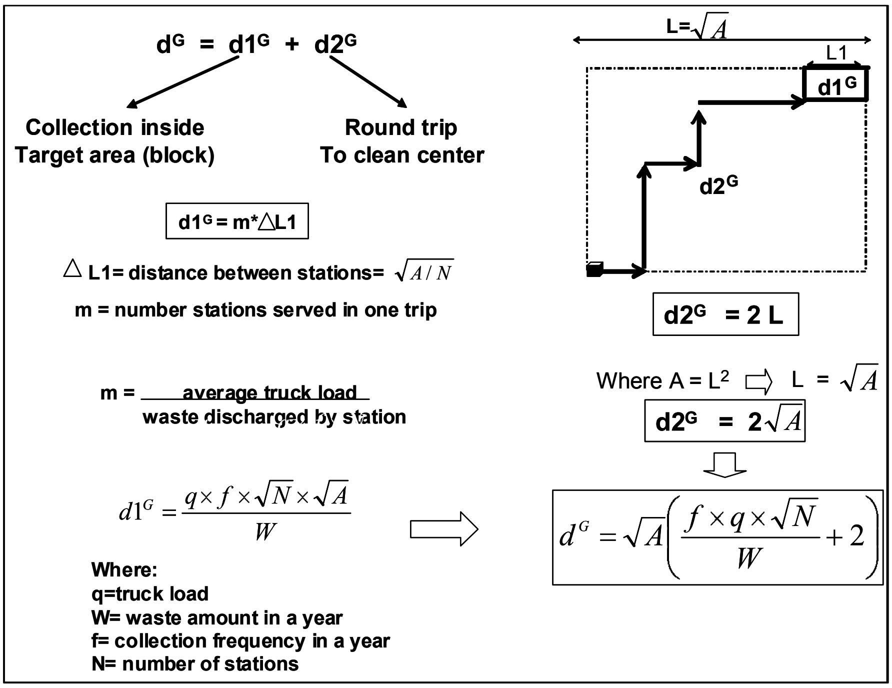

Figure 13.5 Collection analysis: grid city model

The formula shown in Figure 13.5 allows us to calculate the average transport distance of a collection trip (d^G). This means the round trip to the recycling centre or MRF and the distance travelled inside the collection area. Since the total number of collection trips in a year is given by:

$$\text{\# of collection trips/year} = \frac{W}{q} \tag{13.1}$$

It is possible to calculate the total driving distance per year:

$$D\mathrm{G} = \sqrt{A}\left(f \times \sqrt{N} + \frac{2 \times W}{q} \right) \tag{13.2}$$

The actual driving distance differs from the calculated value because the actual roads configuration is not a perfect square, so a tortuosity factor is introduced. This factor is given by:

$$\xi = \frac{D^{\circ}}{\mathrm{DG}} \tag{13.3}$$

In this equation, D^0 denotes the actual observed value in cities for which data on waste transport distances are available. The tortuosity factor is obtained from a correlation graph of the real and calculated distance values for cities with different population sizes. This factor allows a more reliable estimate of the total driving distance per year.

Collection system boundary

The Hyogo eco-town target area includes 4 large cities (80% of total population), 3 medium-size cities, 8 small cities and 42 towns. Our model for big cities assumes a collection model with a Material Recovery Facility (MRF) inside each city. For the rest, we proposed clusters of small and medium-size cities and towns, assuming that the MRF is inside a medium-size city. Details of the Hyogo eco-town area statistics are shown in Table 13.2.

Table 13.2 Hyogo eco-town statistics

Item	Detail
Area	8,391 km^2
Population	5,540,308 people
Domestic waste generation	2,700,000 tons/year
Per capita waste generation	1.33 kg/person.day
Plastic packages waste generation	156,000 tons/year
Per capita plastic waste generation	77 g/person.day
# of cities over 1 million people	1
# of cities over 400,000 people	3
# of cities over 200,000 people	3
# of cities over 50,000 people	8
# of towns with less than 50,000 people	42

Integral plastics recovery in Hyogo eco-town

The objective of applying reverse logistics to plastics recovery in the Hyogo eco-town area is to reduce the collection, pre-treatment and transport costs to make the plastic recycling an economically attractive option. By coupling the reverse logistics system with an adequate recycling technology, using existing industrial facilities, it is possible to construct a wide area plastics recovery system. In order to determine the most adequate recovery technology according to the type of collected waste, it was therefore necessary to analyse the industries that are operating in the eco-town area and possess the technology and know-how that could be used for plastics recovery. The details of the plastics recovery system in Hyogo eco-town are shown in Figure 13.6.

As mentioned above, there is one plastic recovery related technology in Hyogo: the use of plastic wastes in the blast furnace at Kobe Steel Corporation (Kobelco). This furnace uses plastic wastes as a reducing agent for the transformation of iron ore into pig iron. Plastic wastes can also be used in the steel industry as raw materials along with coal in coke ovens. This allows the recovery of coke gas for use as fuel, coke for use in blast furnaces and oil for use in the chemical industry. Besides the possibility of using plastic wastes in the steel industry, the chemical industry and the energy supply industry also possess technologies and know-how that could be used for plastic waste recovery. Gasification is an attractive alternative because the high quality synthetic gas obtained from this process can be used in the chemical industry. Even though all these recovery technologies are already being applied in Japan, this paper only considers the use of plastic wastes as feedstock in blast furnaces as a chemical recycling option besides material and thermal recycling.

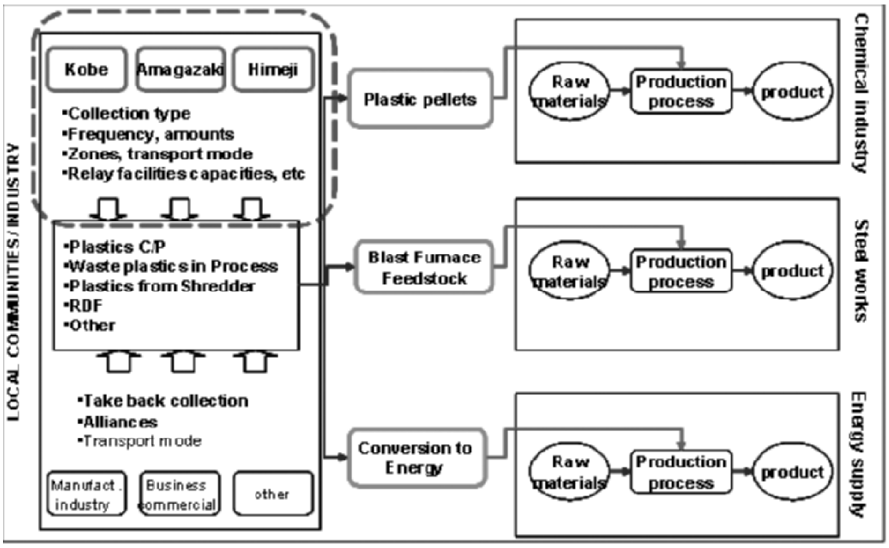

Figure 13.6 System boundaries for plastic wastes recovery

13.4 Eco-efficiency for recycling systems

The concept of eco-efficiency was developed by the World Business Council on Sustainable Development (WBSD) as a way to maximise the economic output or service while at the same time minimising the environmental impacts of human activities. However, the WBSD approach is mostly intended to evaluate the performance of production processes of products and services, and does not include the final recycling and recovery phase (Eik and Brattebo 2001). Since there are many types of recovery systems, it is particularly important to design the eco-efficiency indicators in such a way that they are applicable to the actual system being analysed. Eik and Brattebo (2001) proposed a set of indicators to be developed and applied when analysing an existing or possible future recovery scenario (see Table 13.3). The generally applicable indicators are to be developed for the analysis of all recovery systems. Additionally, if more information is needed, system-specific indicators for the actual system being analysed should be developed and applied. The application of these two types of indicators assesses the eco-efficiency of existing and future recovery scenarios.

Table 13.3 Eco-efficiency indicators for recovery systems

	System analyzed	Decision makers/users	Criteria for the indicators	Indicators
Generally applicable indicators (GAI)	- Entire recovery system - Indicators calculated per functional unit	- National authorities - Local authorities - Recycling companies - Firms	- Valid for all recovery systems - Reflect global environmental concern - Relevant and scientifically valid	- Amount recycled - Resources saving - Energy consumption - CO_2 emissions - Total net costs
System specific indicators (SSI)	- Entire recovery system - Indicators calculated per functional unit	- National authorities - Local authorities - Recycling companies - Firms	- Developed for the actual system - Reflect concern in the system - Relevant and scientifically valid	- Landfill pressure - SO_x - NO_x - etc

Scenario setting

The most important factor in setting our scenarios, along with the Japanese environmental legislation, was the diversion of plastic wastes from landfill, rather than the choice of a specific recovery option. Previous studies had shown that material recycling over 15% has no major benefit in terms of eco-efficiency (Eggels et al. 2001). In Japan, only 7% (PWMI 2004) of plastic wastes currently goes to open-loop material recycling. In the case of Hyogo, this recycling rate is actually zero, mainly due to the imbalance between collectable plastic wastes and potential end markets. Therefore, a maximum of 10% material recycling was assumed. Another important factor was the policy on plastic wastes in Japan, as the Japanese government has set a target of 40% for material and/or chemical recycling of plastic wastes by 2010. Finally, we also considered the recovery technology capacities available in the Hyogo eco-town area in our scenario setting. The resulting scenarios are shown in Figure 13.7.

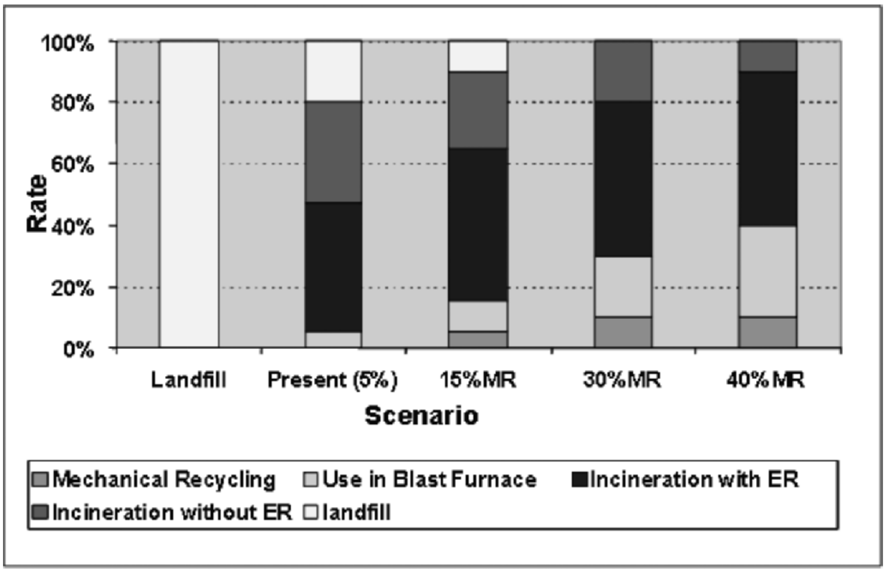

Figure 13.7 Scenarios for plastic wastes recovery

Environmental and cost analysis of the scenarios

The environmental evaluation used the same indicators used for the recycling technologies analysis: global warming potential, resource depletion, energy depletion and landfill use. The economic impact analysis included the costs along the reverse supply chain. The last three scenarios were evaluated with and without reverse logistics.

The normalisation of the environmental impacts was based on the annual emissions in Japan. The values were weighted using the social coefficient developed by the Fraunhofer Institute and the Association of Plastics Manufacturers in Europe (APME 2001). According to this method, the social coefficient for greenhouse gases is 35%, the social coefficient for energy depletion and that for resource depletion are both 25% and the social coefficient for landfill use is 15%. Figure 13.8 summarises the environmental evaluation approach.

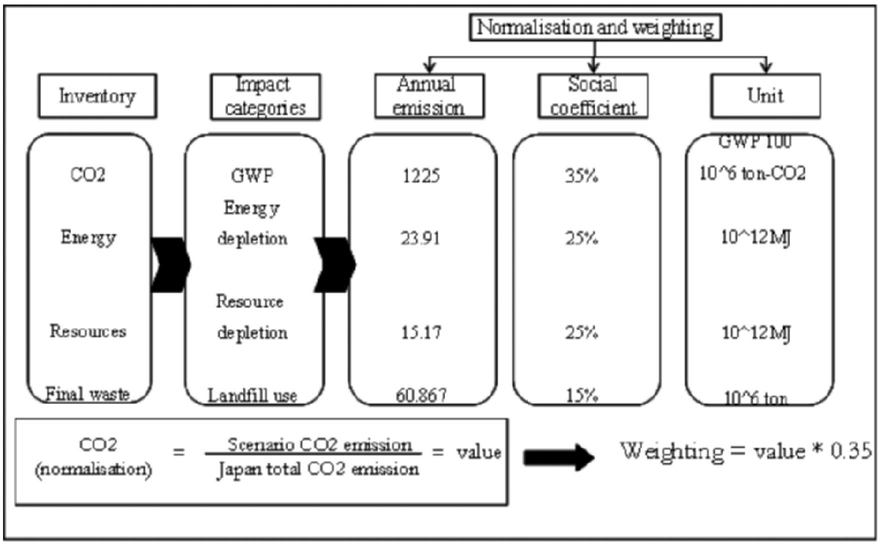

Figure 13.8 Normalisation and weighting of environmental impacts

Eco-efficiency evaluation of the scenarios

The environmental and economic impacts of the scenarios considered for the evaluation of the Hyogo eco-town plastics recovery system were normalised in an eco-efficiency graph (Saling et al. 2002), which is shown in Figure 13.9. The environmental impacts are shown on the vertical axis and the economic impacts on the horizontal axis. The introduction of reverse logistics reduces the collection costs by around 10% in the proposed scenarios (as indicated by the black arrows).

The results of the eco-efficiency analysis allow the following conclusions:

- While 100% landfill is the cheapest option, it has the greatest environmental impact.

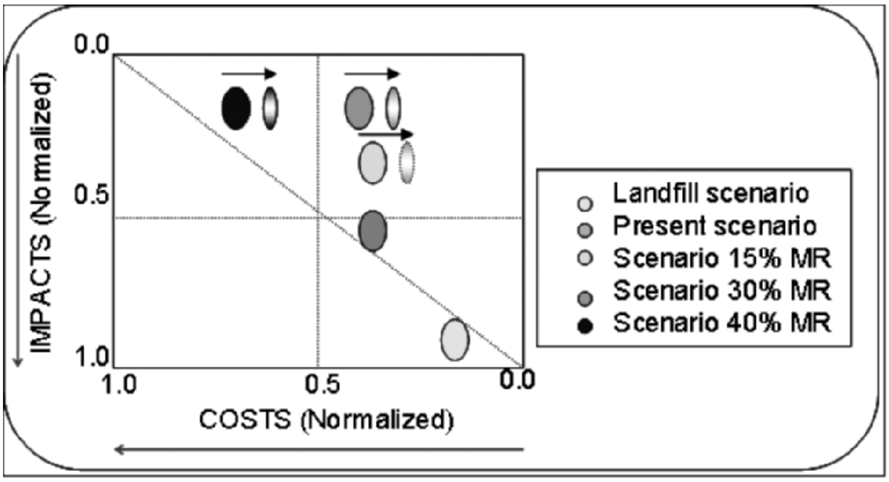

Figure 13.9. Eco-efficiency evaluation of the scenarios

- o The scenario with 40% materials recovery has the best environmental performance but it is at the same time the most expensive option.
- o The scenario that combines 30% materials recovery (10% material recycling and 20% chemical recycling) with a high level of energy recovery (50%) is slightly more expensive than the current situation but offers the best environmental performance. Applying reverse logistics improves the benefit in both respects.
- o The scenario that combines 15% materials recovery (5% mechanical recycling and 10% chemical recycling) with a high level of energy recovery (50%) also offers a better option in terms of both environmental and cost impact than the current situation. Applying reverse logistics greatly improves the benefit.

13.5 Conclusion

This study analysed the possibilities of implementing a reverse logistics system for plastics recovery in the Hyogo eco-town project and its coupling with technologies and know-how available in the area. Applying the reverse logistics concept in the eco-town can produce both environmental and economic benefits by increasing recycling rates and reducing environmental impacts without incurring excessive costs. Our study focused on the environmental and economic performance of the main recovery technologies as well as a detailed evaluation of the plastic waste flow by analysing alternative scenarios to improve plastic waste management in the eco-town. The results allow the following conclusions:

- By using the existing industries and facilities for plastics recovery in Hyogo eco-town and coupling those with an efficient reverse logistics network for plastic collection and treatment it is possible to construct a wide area plastic recycling system.
- It is possible to reduce the overall costs impact by around 10% by introducing a reverse logistics system for plastics collection in the eco-town area.
- Combining increased material recycling levels with efficient reverse logistics allows both environmental and economic gains. However, material recycling levels over 30% seem to produce minimum environmental gain compared with the current approach, while greatly increasing costs: it is necessary to focus on diversion from landfill and simple burning rather than on a specific recovery option.
- Further research should include other sources of plastic wastes, i.e. industrial and commercial sources. The emphasis should also be on other types of recovery technology, especially gasification and pyrolysis (coke ovens).

Note.
Ishikawa has also developed a model to calculate the total number of trucks needed per recycling centre. This formula is important to estimate the economic impacts of the collection model. For more details on this model, see Ishikawa (1996).

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