Sustainable Design Series of Delft University of Technology

LCA-based assessment of sustainability: the Eco-costs/Value Ratio

EVR

Joost G. Vogtländer

Co-authored by B. Baetens, A. Bijma, E. Brandjes, E. Lindeijer, M. Segers, F. Witte, J.C. Brezet, Ch.F. Hendriks

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The Eco-costs/Value Ratio (EVR)

Original publications on the theory,
updated with eco-costs 2007 data

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Preface and Acknowledgements

The ever growing economy seems to be one of the major root-causes of the continuing deterioration of our environment. The question is: what can be done? Stopping the economic growth seems no realistic option, so the solution must be found in a better eco-efficiency of our systems for production and consumption ("de-linking of economy and ecology").
Future products and services need to have a high value/costs ratio combined with a low burden for our environment. This is the challenge for modern designers, engineers, business management and governmental leaders.

This book is on the basic aspects of the Model of the Ecocosts/Value Ratio, an LCA based Decision Support Tool on the sustainability of products and services. It is a compilation of the original publications in scientific journals (peer reviewed), and some additional issues of the Doctorate Thesis which were not published in journals.

After the first set of publications on the eco-costs in the period 1999–2004, the system of the eco-costs has been renewed, resulting in a new dataset: the eco-costs 2007, based on new characterization tables for more then 3000 emissions, and based on a new curve of the marginal prevention costs of summer smog ('photochemical oxidation' or 'respiratory organics') and a new assessment of carciogens. The marginal prevention costs of the other 'midpoints' were checked and corrected for monetary inflation of costs.
For the convenience of the reader, the tables and the numbers in the text have been updated accordingly.

Acknowledgements

There are so many people who contributed to the model, that it is not feasible to name them all. Some people, however, did more than only comment the ideas, but contributed to specific issues of the model: Bianca Baetens (recycling of construction materials for buildings), Arianne Bijma (the issue of communication), Eduard Brandjes (the transport case), Dolf Gielen (eco-costs of energy), Erwin Lindeijer (land-use), Merel Segers (eco-costs 2007 calculations) and Flip Witte (botanical value and eco-costs of land).
I would like to express my gratitude for the valuable contribution of prof.dr.ir. J.C. Brezet and prof.dr.ir. Ch.F. Hendriks †. Without their contribution, the development of the EVR model would not have succeeded.

Delft University of Technology, the Netherlands, November 2009
Joost G. Vogtländer

Preamble

Prosperity: a fragile balance between economy and nature

On Venice, 1974:

" . . . . . . Almost every winter for many years, large parts of the city have become flooded. Indeed, this is becoming an even more frequent occurrence. It is due to the subsidence of the entire area under and around the lagoon, which in turn has been caused by the abstraction of groundwater by industry and agriculture in the surrounding region. … The rising local seawater level has caused damp in the walls of many buildings, which has damaged many paintings and frescos. Air pollution, caused by a chemical industry which is not adequately supervised, has caused irreparable damage to sculptures and buildings. Much has already been lost and unless action is taken soon, at least half of the art treasures which remain will also be lost within the next forty years. … The problems faced by Venice are primarily of a social nature. Tourism does not provide sufficient revenue for the winter months. Young people prefer to live on the mainland, where they can have their own car parked outside the front door rather than having to walk or rely on boats. Houses in Venice itself are rapidly decaying. New sources of revenue must therefore be found in order to make the old city an attractive place to live in once more … "

From: *Grote Winkler Prins Encyclopaedia*, seventh edition, 1974, (in translation).

It is with some hesitation that I selected the above to serve as the introduction to this book. Is it relevant to the topic of sustainability and eco-efficiency? Is the picture presented a realistic one? Can the same phenomenon, or one broadly similar, also be seen elsewhere?

The situation described presents many facets of the same reality. However, the significant characteristic is that it is impoverishment which is leading to decay: there are insufficient funds for maintenance, let alone for new measures such as the construction of a drainage system. Faced with the threat of greater unemployment, the government allows industry and agriculture to place an unwarranted burden on the local environment. (This is a dilemma we have seen not only in Eastern European countries and the developing countries, but also in the Netherlands. Here too, numerous instances can be cited in which the government has succumbed to pressure from various business lobby groups and has failed to take appropriate measures, resulting in harm to the environment).

For Venice, the prospects are now more encouraging than was the case twenty years ago:

- the Italian government has now prohibited any further abstraction of water by industry
- the historic city centre is being refurbished with international assistance

- new economic activity is being developed in the service sector, located in the city centre.

The new challenge, however, is to withstand the ever growing mass of tourists who are attracted by inexpensive travel arrangements, and to withstand the increased frequency of flooding.

The policy to be adopted is clear: the city can only survive if it has sufficient economic strength (i.e. ongoing prosperity) to be able to stop the ecologically harmful activities, construct sewers, and perhaps construct a seawater barrier which is normally open but can be closed at high tides.
At the same time, strong economic growth must not itself result in any additional environmental impact (e.g. de-linking of economy and ecology is the key to a sustainable development)

It would seem that in our modern world, the concept of 'sustainability' has become quite complex. It now goes far beyond the encouragement of an alternative 'simpler' lifestyle (Dutch: ‘consuminderen’), as is illustrated by the anecdote on Diogenes:
when Alexander the Great promised him anything whatsoever he might desire, Diogenes merely asked Alexander to stand aside, out of the sun.

Palazzo Capello Malipiero, La Volta del Canal, Venice: ‘Water is a boon in the desert, but the drowning man curses it’ (English proverb).

Contents

1 Introduction

1.1 Purpose of this book

The primary purpose of this book is to provide students, and other people who are interested in the subject of sustainablility, with theoretical background information on the eco-costs system and the model of the Eco-costs/Value Ratio (EVR).[1]

Eco-costs is a measure to express the amount of environmental burden of a product on the basis of prevention of that burden. It are the costs which should be made to reduce the environmental pollution and materials depletion in our world to a level which is in line with the carrying capacity of our earth.
For example: for each 1000 kg CO_2 emission, one should invest € 135.– in offshore windmill parks (and other CO_2 reduction systems at that price or less). When this is done consequently, the total CO_2 emissions in the world will be reduced by 65% compared to the emissions in 2008. As a result global warming will stabilize. In short: "the eco-costs of 1000 kg CO_2 are € 135.–".
Similar calculations can be made on the environmental burden of acidification, eutrification, summer smog, fine dust, eco-toxicity, and the use of metals, fossil fuels and land (nature). As such, the eco-costs are virtual costs, since they are not yet integrated in the real life costs of current production chains. The eco-costs should be regarded as hidden obligations.
The eco-costs of a product are the sum of all eco-costs of emissions and use of materials and energy during the life cycle "from cradle to cradle". Eco-costs calculations are based on Life Cycle Assessment (LCA), as defined in ISO 14040 and 14044.
The practical use of eco-costs is to compare the sustainability of several product types with the same functionality. The advantage of eco-costs is that they are expressed in a standardized monetary value (€) which appears to be easily understood 'by instinct'. The calculation is transparent and relatively easy, compared to damage based models.

The EVR is a so-called E/E indicator ("Ecology/Economy Indicator") which can be applied in cases where a designer (architect, product engineer, marketing manager, etc.) is asked to design a product (a house, a road, an appliance, a service, etc.) within a given

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[1] For specialists it is often not easy to understand the eco-costs system and the model of the EVR. The main reason for this is that it requires a fundamental paradigm shift to make the step from 'damage based' systems (which are common in LCA) to 'prevention based' systems. On paradigm shifts, Edward de Bono said: "you cannot see what your mind is not prepared for".

price (budget). The issue then is to create maximum value for the end-user at a minimum of eco-costs (environmental burden). We call this 'ecoefficient value creation'.
The EVR model can not only be applied in the stage where the design is ready (the classic LCA approach), but can also be applied in the early design stages of feasibility studies (when data on costs and market values are estimated). Calculations in combination with LCC and WLC are possible as well.

The rather complex issue of 'allocation' in Product-Service Systems and in the End of Life phase, has been resolved in a practical and consistent way (where the existing LCA methodology failed until now to provide sufficient practical answers). This makes the system suitable for Cradle to Cradle calculations.
Furthermore the model comprises a system for modelling the issue of land-use, to be able to facilitate decisions with regard to spatial planning.
The theoretical basis of the model has been introduced in 1999, and published in 2000-2004 in the International Journal of LCA (Vogtländer, Bijma, 2000, Vogtländer, Brezet, Hendiks, 2001,B, Vogtländer, Hendriks, Brezet, 2001,C)and also in the Journal of Cleaner Production (Vogtländer, Bijma, Brezet, 2002, Vogtländer, Lindeijer, Witte, Hendiks, 2004). This book is a compilation of these publications and some important additional issues from the Doctorate Thesis on the EVR (Vogtländer, 2001,A).
For the convenience of the reader, the tables and the numbers in the text have been updated according to the new set of data, the eco-costs 2007, so that information in the publication has become in line with data which are provided in the other books of the Sustainable Design Series of the Delft University of Technology and the website www.ecocostsvalue.com.

1.2 Mission

In November 1993, the World Council for Sustainable Development (WBCSD) defined eco-efficiency as:

> "the delivery of competitively priced goods and services that satisfy human needs and bring quality of life, while progressively reducing ecological impacts and resource intensity, throughout the life cycle, to a level at least in line with the earth's estimated carrying capacity."

This business oriented definition links two aspects of good governance:

- Modern management practice ("*the delivery of competitively priced goods and services … quality of life").*
- The need of a sustainable society *("while progressively reducing … to … earth's carrying capacity"*).

The first part of the sentence asks for a maximum value/costs ratio of the business chain, the second part of the sentence requires that this is achieved at a minimum level of ecological impact. But what does this rather philosophical definition mean to business managers, designers and engineers in terms of the practical decisions they take?

There is a need to resolve simple questions like: what is the best product design in terms of ecological impact?, what is the best product portfolio in terms of sustainability?, what is the best sustainable strategy?

These issues are also related to the Triple P concept of the triple 'bottom line' as formulated by John Elkington (Elkington, 1998). In corporate decision taking, equal weight should be given to the following three aspects:

- 'People', the social consequences of the total Life Cycle
- 'Planet', the ecological consequences
- 'Profit', the economic profitability (being the source of 'Prosperity')

The EVR model unravels the system of the 3 P's, primarily analysing carefully the P of Prosperity (value) and the P of Planet (eco-costs), and analysing the interaction of these 2 P's in the total system. See Figure 1.1.

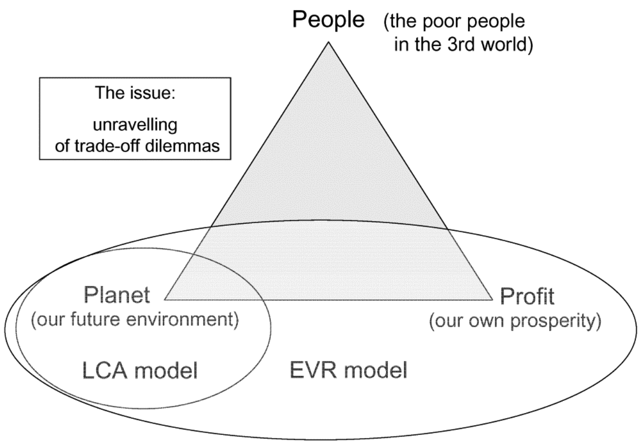


Figure 1.1. The EVR model is about 2 P's of the triple P model.

The third P, the P of People (of the developing world) is of an extreme complex nature, but related to eco-efficiency as well. The need for a better organized economy, de-linking the economic growth and the environmental degradation, was expressed for the first time in the Brundtland Report 'Our Common Future' (1987, page xii, see also Appendix 1), as the conclusion of a study on the situation in the developing countries:

> "The downward spiral of poverty and environmental degradation is waste of opportunities and of resources. In particular it is a waste of human resources. These links between poverty, inequality, and environmental degradation formed a major theme in our analysis and recommendations. What is needed now is a new era of economic growth - growth that is forceful and at the same time socially and environmentally sustainable."

The issue is how to translate the above mentioned missions of creating a sustainable society to a practical tool for designers, engineers and architects. One of the key aspects of the required de-linking of economy and ecology is the fact that products and services need to have a low ratio of their eco-costs and their value (EVR).

1.3 Eco-costs 2007, a single indicator for LCA

The eco-costs method is used in LCIA to express the amount of environmental burden of a product or service, on the basis of prevention of that burden. Eco-costs are the costs which should be made to reduce the environmental pollution and material depletion in our economy to a level which is in line with the carrying capacity of our earth (the so-called 'no-effect level'). As such, the eco-costs are virtual costs, since they are not yet integrated in the real life costs of current production chains (Life Cycle Costs). The eco-costs should be regarded as hidden obligations.

The eco-costs of products are based on the sum of the marginal prevention costs ('end of pipe" as well as system integrated) during the life cycle (cradle to grave as well as cradle to cradle) for toxic emissions, material depletion, energy consumption and conversion of land. The structure of the calculation system is depicted in Figure 1.2. The advantage of eco-costs is that they are expressed in a standardized monetary value (€) which appears to be easily understood 'by instinct'. The calculation is transparent and relatively easy, compared to damage based models which have the disadvantage of extremely complex calculations with subjective weighting of the various aspects contributing to the overall environmental burden (Bengtsson and Steen, 2000, Finnveden , 2000).

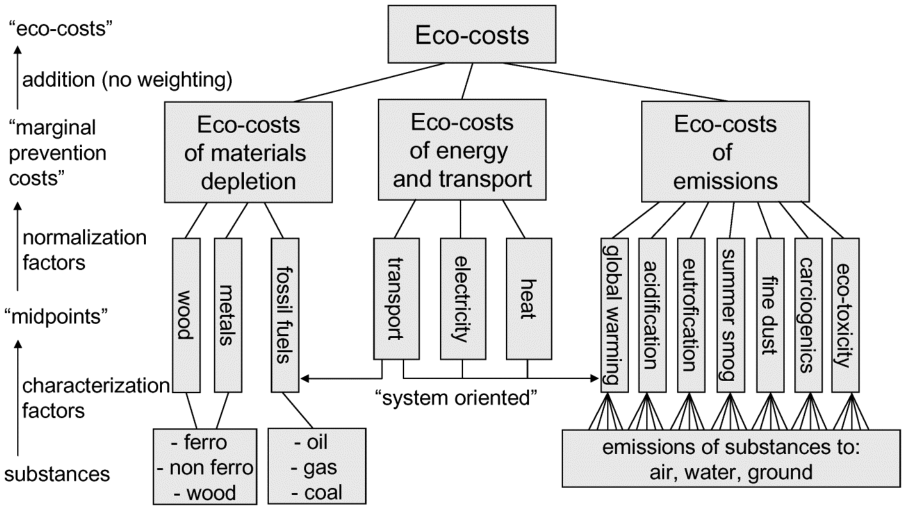


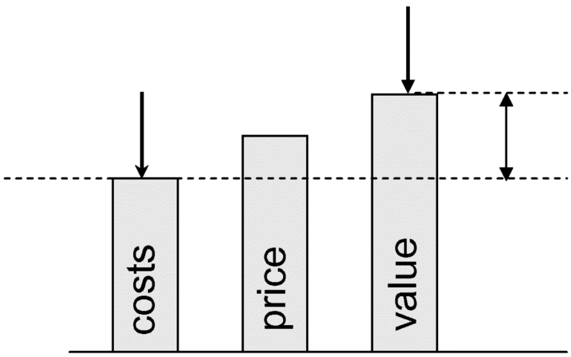
Figure 1.2. Calculation structure of the eco-costs 2007.

The method of the eco-costs 2007 comprises tables of over 3000 emissions, and has been made operational by special database for Simapro, based on LCIs from Ecoinvent v2 and Idemat 2008 (over 5000 materials and processes), and a database for CES (Cambridge Engineering Selector). Excel look-up tables are provided at www.ecocostsvalue.com.

Note. Prevention measures will decrease the costs of the damage, related to environmental pollution, e.g. damage costs related to human health problems (Holland, Watkiss, 2003). The savings which are a result of the prevention measures are of the same order of magnitude as the costs of prevention. So the total effect of prevention measures on our society is that it results in a better environment at virtually no extra costs, since costs of prevention and costs of savings will level out.

1.4 Perceived Customer Value

To understand the EVR model, and to understand the de-linking of economy and ecology, it is essential to understand the concept of 'perceived customer value'[2] in modern management. Each product and each service has 3 economic dimensions: the costs, the price and the socio-economic (market) value. See Figure 1.3. These dimensions have all money (e.g. €, $, etc.) as unit, but must strictly be kept separate (it is obvious that adding components of the cost to the price has no practical meaning at all; the same applies to the value).



Note: value = product quality + service quality + image

Figure 1.3. The costs, the price and the value of a product or service.

In the modern management approach, the strategic focus is on the ratio of value and costs. The value is normally a bit higher than the price ('a buyers market'), but might also be a bit lower than the price ('a sellers market'). In the EVR model we take the

2 'perceived customer value' might be defined as "the use and fun which is expected after the purchase, as seen through the eyes of the customer"

average case where the value is the 'fair price', which is the price which the average buyer in the specific market niche is prepared to pay.

In the classical management paradigm, higher value (´quality´) leads always to higher costs. In the modern management paradigm that is not the case: there are many management techniques that lead to a better value/costs ratio. Examples are: logistics (better delivery at lower stock levels), complaint management (satisfied customers with less claims), waste and quality management (less materials, better quality). All these examples – there are many more in the field of Total Quality Management and Continuous Improvement - lead to more value at less costs. This is called 'the double objective' for managers and opens new perspectives to support eco-efficiency (it supports the first part of the eco-efficiency definition of the WBCSD). Note that this modern management philosophy is much more than just 'adding services' to existing products. It is about carefully improving the quality of products and services (as perceived by the customer) by eliminating the 'non value added' energy, materials and work.

A fact is that these modern management techniques not always lead to better eco-efficiency (e.g. the use of pesticides in agriculture results in a better value/costs ratio but not in a better level of environmental protection). That is why the aforementioned definition of eco-efficiency of the WBCSD adds *"… while progressively reducing ecological impacts …"*.

For this reason, companies which aim at good governance must make sure that their products have low eco-costs. LCA is here an indispensable tool.

More information on the dynamic aspects of perceived customer are given in Appendices 5, 6 and 7.

1.5 The Ecocosts-Value Ratio (EVR)

The Ecocosts/Value Ratio, EVR, is an indicator which fulfills 3 different functions:

1. It is an indicator for sustainability in LCA (additional to the eco-costs) in cases where the quality of products (with the same functionality) differs.
2. It is an indicator which is relevant to corporate strategies and governmental policies: it links the consumer side with the production side (see Chapter 5).
3. It is a parameter in the so-called *economic allocation* of LCA calculations (see Section 3.5).

The aim of an LCA is often to compare two products (or services). A prerequisite is then that the two products have the same functionality *and the same quality* (in the broad sense of the word).

In practice, however, new innovative 'green' designs often have the same functionality, but differ from the classical design. In such cases the quality is not the same. It is a widespread misunderstanding that the design with the lowest eco-costs (or millipoints, or carbon footprint) is always the best choice in terms of sustainability. When the eco-

costs of the new design are lower and the quality is better, there is no doubt that the new design is more sustainable. However, when the quality of the new design is lower, it remains to be seen which design alternative is the best choice in terms of sustainability.
In cases where the quality differs, the Ecocosts/Value Ratio, EVR, appears to be a better indicator for sustainability. This is because "value" (fair price) is a good indicator for the quality in the broad sense.

The EVR is a so-called E/E indicator, which means that it is an indicator to describe the eco-efficiency of a product and/or service. The EVR is a dimensionless number which indicates to what extent a (design of a) product contributes to the de-linking of economy and ecology. Most of the other E/E indicators which are proposed in literature, divide eco-burden by costs (or the other way around). The EVR, however divides the eco-costs by customer value, which brings the customer behaviour into the equation.
In the model of the EVR, a product (and service) has 3 separate dimensions: the costs, the eco-costs and the value. See Figure 1.4. These dimensions have all a monetary unit (e.g. €, $, etc.), but must strictly be kept separate (it is obvious that adding components of the cost to the value has no practical meaning at all; the same applies to the eco-costs).

There is a consumer's side of the de-linking of economy and ecology. Under the assumption that most of the households spend in their life what they earn in their life, the total EVR of the spending of households is the key towards sustainability. Only when this total EVR of the spending gets lower, the eco-costs related to the total spending can be reduced even at a higher level of spending. There are two ways of achieving this:

1. at the production side: the improvement of eco-efficiency ('lowering EVR') of products and services by the industry
2. at the consumer's side: the change of lifestyle of customers in the direction of 'low EVR' products.

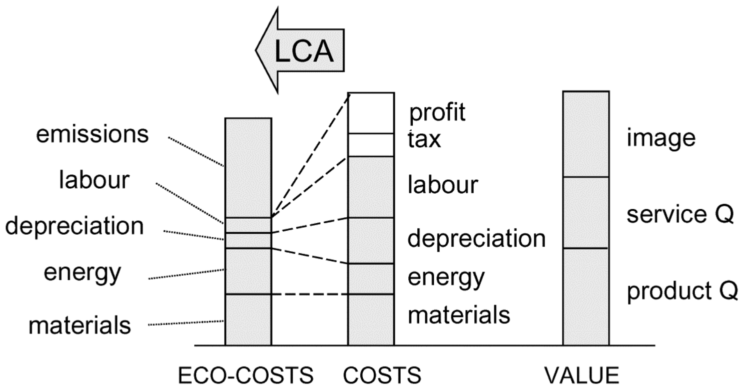


Figure 1.4. The value, the costs and the eco-costs of a product and/or service.

The EVR can also act as a parameter for economic allocation in LCA calculations, especially for services (eco-costs per € instead of eco-costs per kg). The issue is that services are characterized by shared use of facilities (for transport, offices, equipment etc.) which is complicating the LCA, since materials and emissions are shared as well. Materials and emissions must then be allocated to a specific service in line with the economic importance of that specific service, the so-called 'economic allocation' in LCA.

1.6 A new data set: the eco-costs 2007

The original eco-costs 1999 were based on characterisation tables of the eco-indicator 95 and prevention costs of RIVM of 1997 (Delink and Van der Woerd, 1997).
After the first set of publications, the basic data have been discussed extensively, and were adapted to new studies and tables from literature.
The characterisation tables in the eco-costs 2007 system are:

- IPPC 2007, 100 years, for greenhouse gases
- CML-2, for acidification, eutrification and summer smog (photochemical oxidation)
- IMPACT 2002+, for aquatic eco-toxicity (inc. heavy metals), fine dust (was winter smog) and carcinogens

Although calculations on marginal prevention costs only change with monetary inflation (see Appendix 3), the calculation on the prevention costs of summer smog has been revised entirely since new data came available (Cronenberg, 2000), and since two effects influenced the calculations considerably: the innovations in water based paint systems and the innovations in motor management in the automotive industry. Both innovations resulted in a drastic change of the curve of prevention costs, and therefore a drastic change in the marginal prevention costs. The marginal prevention costs of carcinogens has been changed as well, based on the aforementioned study on summer smog.
The calculations of prevention costs of greenhouse gases of ECN were checked with an extensive study on the costs of wind parks at the sea[3] by the University of Leuven (Van Capellen, 2005), but there was no need for a change other than the monetary inflation.
Note. Aquatic eco-toxicity (including heavy metals), fine dust, and carcinogens are rather problematic in LCA, since there effects are non-linear (LCA is inherently a linear calculation system) and often specific for the typical local situation. See also Appendix 2. These emissions, however, are kept within the eco-costs system to maintain the 'signalling function' (showing that the toxicity of the product is OK in the Life Cycle).

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[3] The eco-costs (marginal prevention costs) of greenhouse gases (CO2) are determined by the costs of substitution of electricity from coal fired power plants by electricity of windmill parks at the sea. The reason why will be explained in Chapter 2.

1.7 The structure of this book

This book is a compilation of a series of publications in scientific journals. Exact reference data of the original publication is provided at the first page of each chapter. The text in the book is nearly a verbatim copy of the original text, however, the numbers and tables are new (updated) with respect to the ecocosts 2007.
The advantage of the verbatim versions is that each chapter of this book can be read 'stand-alone'. The disadvantage is that some general information is repeated in each chapter.
The reference lists of literature have been combined.

Since the Doctorate Thesis contains more information than the articles in the scientific journals, some additional information is provided in the Appendices of this book. For each Appendix the reference page(s) of the Doctorate Thesis are provided. Sometimes the text in this book is a verbatim copy of the Thesis, in some Appendices the text of the Thesis has been shortened.

2 The Virtual Pollution Prevention Costs[4]

2.1 Abstract

This chapter deals with the development of a single indicator for toxic emissions.
In literature many models (qualitatively as well as quantitatively) can be found to cope with the problem of communicating results of LCA analyses with decision makers. Most models translate data on emissions in a single indicator, using a classification and characterization step. More than 30 of these models have been looked at, of which 14 have been studied in detail. From these analyses it was concluded that there is still a need for further development.

A new model for a single indicator has been designed on the basis of the following main criteria:

1. The model has to be easy explainable to non-specialists (i.e. the model has to relate to 'normal life')
2. The model has to be 'transparent' for specialists:
   - Since the choice of the region influences all these kinds of calculations, specialists have to be able to adapt the data for the calculation to cope with the choice of a specific region (the data in this publication is for the Dutch and West European region).
   - Since the character of these calculations is that some arbitrary decisions cannot be avoided, the model has to have a structure that enables an easy assessment of the effect of these decisions, so that the experts can adapt the model to their own judgements.

Based on the analyses of the aforementioned existing models, it was concluded that a model based on the marginal prevention costs seems to give the best fit with the two criteria mentioned above.
These marginal prevention costs are assessed for seven emission effect classes on the basis of prevention measures which are based on readily available technologies. The costs of the measures are based on current West European price levels.

[4] The original title was: "The virtual pollution prevention costs '99. A single LCA-based indicator for emissions." Published in Int. J. of LCA (Vogtländer , Bijma, 2000).

Essential to the model is that it has to be judged whether the set of measures is sufficient to reach a sustainable level of emissions.
Given a certain region one can calculate the effect of the set of measures (provided that enough data on that region is available) for the current situation. These calculations, based on West European current price levels, have been made for The Netherlands as a region for the following classes of emissions:

- Acidification, eutrophication, summer smog, winter smog and heavy metals, based on previous work of IVM, Amsterdam
- Global warming by CO_2 emissions based on previous work of ECN, Petten.

Furthermore, it has been checked as to how the assumptions are related to the current emission targets of the Dutch government, and it is discussed how this data may relate to other regions in the world.

The following data set is proposed to be applied as marginal prevention costs:

- prevention of acidification 7.55 €/kg SO_x equivalent
- prevention of eutrophication 3.60 €/kg phosphate equivalent
- prevention of ecotoxicity (heavy metals) 802 €/kg Zn equivalent
- prevention of carcinogens 33 €/kg PAH equivalent
- prevention of summer smog 8.90 €/kg C_2H_4 equivalent
- prevention of fine dust (winter smog) 27.44 €/kg fine dust PM2.5
- prevention of global warming 0.135 €/kg CO_2 equivalent.

The 'virtual pollution prevention costs' is proposed as a single indicator for emissions, being the sum of the marginal prevention costs of all aforementioned classes of pollution.

2.2 The problem of weighting several types of emissions

A generally accepted route towards a single indicator is an approach which is based on splitting the problem into two levels (ISO 14040 and 14044):

1. Combining emissions with the same nature of effect: the so-called 'classification' in groups; followed by weighting of the importance of an emission within each class: the so-called 'characterization' within the group. For each group this leads to an "equivalent weight of the major pollutant in the class".
2. Finding a weighting principle to add up the different classes.

For most of the major pollutants, the classification and the characterization factors (i.e. the weighting factors within classes) can be assessed from the chemical, physical or biological effect they have:

- Acidification: characterized by simple formulas from chemistry
- Eutrophication: characterized by simple formulas from chemistry

- Summer smog: characterized by relative simple chemical reactions which form ozone
- Winter smog: characterized by the "just detectable effect at long term exposure",
- Heavy metals: characterized by the "just detectable effect at long term exposure", norms given by the World Health Organization in the Air Quality Guidelines for Europe and Water Quality Guidelines for Europe[5]
- Carcinogens: derived from the rate of development of cancer (number of patients in a population of 1 billion people)
- Global warming: rather complex calculations on the reflection of light and its thermal consequences (note that only the relative effects of the several gases have to be known for the weighting).

The characterization factors resulting from the above criteria, which are used in the model for pollution prevention costs, are given in Table 2.1. For a full list of more than 3000 emissions, see www.ecocostsvalue.com.
For background information of the calculation, see Appendix 2

pure emissions		Charact. factor	pure emissions		Charact. factor
Global warming (GWP100)			Carciogenics		
CO2	Air	1	PAH	Air	1
N2O	Air	296	Benzo[a]pyrene	Air	10.0
Dichloromethane	Air	10	chloroform	Air	0.0007
HFC-125	Air	3400	chromium	Air	0.034
HFC-134a	Air	1300	dioxins	Air	486.819
HFC-143a	Air	4300	formaldehyde	Air	0.0003
HFC-152a	Air	120	oxazepam	Air	0.0037
CFC-12	Air	10600	PAH	Air	1
Methane	Air	23	styrene	Air	3,7E-05
Trichloromethane	Air	30			
			Summer Smog		
Acidification			CxHy	Air	0.398
SO2	Air	1	1,1,1-trichoroethane	Air	0.021
HCL	Air	0.88	Acetaldehyde	Air	0.641
HS	Air	1,88	Acetone	Air	0.094
HF	Air	1,6	Alcohols (non specified)	Air	0.356
HNO3	Air	0.51	Aldehydes	Air	0.657
Ammonia	Air	1,88	Benzaldehyde		-0.092
NO	Air	1,07	Benzene	Air	0.218
NO2	Air	0.7	Butadiene	Air	0.851
Nox	Air	0.7	Butane	Air	0.352

Table 2.1. A summary of Characterization factors, mass based; eco-costs 2007 system.

[5] In the eco-costs 2007 system, the heavy metals are in the class of aquatic eco-toxicity and not part of any table on human health (as a result of discussions with CLM, University of Leiden).

	Air	0.8
SO4	Air	0.8
Eutrophication		
Nox	Air	0.13
Ammonia	Air	0.35
NO	Air	0.2
NO2	Air	0.13
Phosphate	Air	1
Nitrates	Air	0.1
COD	Water	0.022
NH3	Water	0.33
Phosphate	Water	1
NH4+	Water	0.33
Ptot	Water	3,06
Ntot	Water	0.42
Aquatic Ecotoxicity with Heavy Metals		
Zn	Water	1
Heavy metals	Water	1
Al	Water	0
Ba	Water	0.057
Cd	Water	2,08
Co	Water	2,76
Cu	Water	0
Fe	Water	0
Pb	Water	0.19
Mn	Water	0.00
Hg	Water	11,26
Ni	Water	0.90
Se	Water	2,43
Ag	Water	0.00
Fine Dust (Winter Smog)		
Fine Dust (PM2,5)	Air	1
SO2	Air	0.535
Carbon black	Air	0.535
Heavy soot	Air	0.535
Iron dust	Air	0.157
dust from building indus.	Air	0.157
mechanical dust	Air	0.157
dust from fires	Air	1
dust from diesel enigines	Air	1
dust from industrial combustion	Air	1

CO	Air	0.027
Crude oil	Air	0.398
CxHy aliphatic	Air	0.352
CxHy aromatic	Air	0.985
Cyclohexane	Air	0.29
Cyclohexanol	Air	0.518
Decane	Air	0.384
Diacetone alcohol	Air	0.307
Diethyl ether	Air	0.445
Diethyl ketone	Air	0.414
Ethane	Air	0.123
Ethanol	Air	0.399
Ethene (C2H4)	Air	1
Ethylene glycol	Air	0.373
Ethyne	Air	0.085
Formaldehyde	Air	0.519
Heptane	Air	0.494
Hexane	Air	0.482
Isobutane	Air	0.307
Isobutanol	Air	0.36
Isobutene	Air	0.627
Isopentane	Air	0.405
Isoprene	Air	1,092
Methane	Air	0.007
Methanol		0.14
Methyl ethyl ketone	Air	0.373
Methyl formate	Air	0.027
m-Xylene	Air	1,108
Non methane VOC	Air	0.6
PAH	Air	0.761
Nonane	Air	0.414
Octane	Air	0.453
o-Xylene	Air	1,053
Pentanal	Air	0.765
Pentane	Air	0.395
Petrol	Air	0.398
Propane	Air	0.176
Propene	Air	1,123
Propanal	Air	0.798
Styrene	Air	0.142
Terpentine	Air	0.377
Propylene glycol		0.457
Toluene	Air	0.637
Undecane	Air	0.384
Vinylchloride	Air	0.021
VOC	Air	0.398
p-Xylene	Air	1.01

2.3 Weighting principles for the different classes

The chemical, physical and biological characteristics of each class differs, so a weighting principle has to be found to add up these different classes.

In general, there are 3 ways to weigh several different types of *potential* damage:
1. weigh the negative value of the damage (the impact)
2. weigh the required effort to prevent the damage
3. weigh the required effort to repair the damage.

It is generally accepted that the third option is in general not the desired option for sustainability problems, since repair of emissions is either not possible or much more expensive than prevention.
So we can weigh the classes either according to type 1 (impact) or type 2 (prevention).

In general, it is possible to weigh either impact or prevention by:
1. 'points'
2. 'money'.

The four resulting possibilities for weighting are depicted in Figure 2.1.

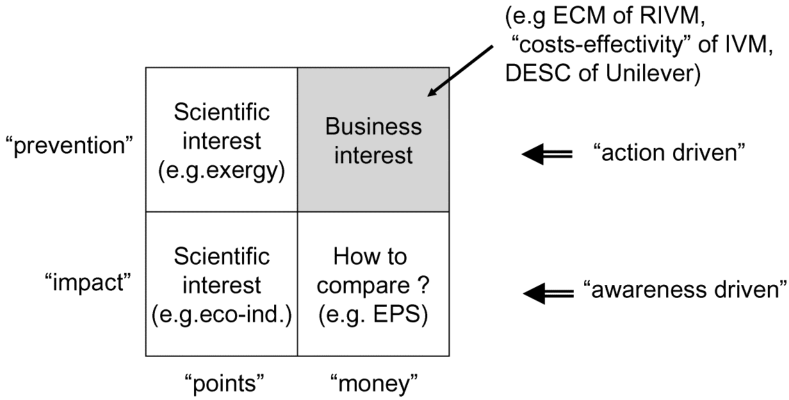


Figure 2.1. Portfolio of models for a single indicator.[6]

The vast majority of the models for a single indicator are based on the combination of 'impact' and 'points', perhaps a result of the fact that environmentalists often use LCAs to make other people aware of the gloomy problem (the potential damage or impact).
The Swedish EPS model is based on 'willingness to pay' which is determined by assessing the negative value of the damage (impact), so this system is a combination of impact and money.
In the exergy models, which are currently being developed, calculations are made on

6 General literature on description of tools: (Braunschweig, 1996, Tulenheimo, 1996) (Graedel, 1998) Hoogendoorn, 1998, Nijland, 1998, Haas, 1997, Beetstra, 1998, Müller, 1997).

the prevention of emissions, so these calculations are a combination of 'prevention' and 'points'.

There are 2 macro-economic models which are not LCA based but which are basically a combination of 'prevention' and 'money': the Milieu Kosten Model (Environmental Costs Model) of RIVM, Bilthoven, and cost effectivity model of IVM, Amsterdam. The DESC model of Unilever is designed for micro-economic decisions (choices on products and processes) and also belongs to this category.
Damage based models have to be used to convince a group of people that environmental protection is something which has to be taken very seriously, and any further damage to our earth has to be stopped.
Models for weighting based on damage (impact), however, have two fundamental problems:

1. Weighting of the impact is a very subjective and arbitrary matter: how to compare a fatal illness with dying trees and/or extinguishing species?? (Finnveden, 1997, Finnveden, 2000, Bengtsson, 2000)
2. An assumption in damage based models is that the damage is proportional to the concentration and to the emissions, which is far from reality.

Prevention based models are to be used when it is accepted by a group of people that pollution and depletion have to be stopped. In such a situation, analyses are required on the way how to prevent. Prevention based models can help to analyse which is the most efficient and effective road towards a sustainable society, rather than accept the damage. Weighting on the basis of impact (damage) is the wrong approach then, since the focus is on prevention and the required strategies for the future.

Models for weighting based on prevention all suffer from the problem of setting the sustainable norms for emissions; basically there are three types of norms:

1. the absolute norms for maximum emissions at the sustainable level,
2. the norms based on the economic optimum of prevention: the emission level where the costs of prevention equal the costs of damage (impact), see Figure 2.2,
3. the current practice of prevention, being the BAT (Best Available Technology) or the 'revealed preference' (Huppes, 1997); note that this 'revealed preference' is not used here for target setting, but only for weighting of the relative importance of each of the classes.

The first method (1.) suffers from the fact that, from a scientific point of view, it is not possible to predict these "absolute" norms (is the complex calculation method 100% correct? How can it cope with all future developments and risks?).

The second method (2.) suffers basically from the same problem as the damage based methods: how to quantify the value of the damage caused by the emissions.

Note. In a qualitative form, however, this method in some specific cases plays a role in decision taking. An example is the ban on CFCs within the EC: it was obvious that prevention costs of using other gases were much lower than the damage costs of the damage to the ozone layer.

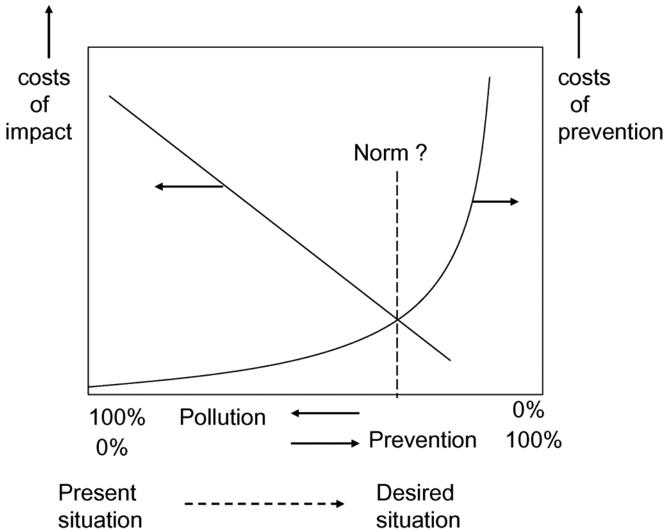


Figure 2..2. The economic optimum as a norm for emission prevention?

The last method (3.) is widely applied within the EC (the IPPC-directive). The key question here, however, is what is affordable in a free market.

The methodology of the 'revealed preference' suffers from the fact that pollution and depletion is not something which is traded on free and transparent markets. Valuation of 'non market' and 'non use' issues is hardly possible (Henley, 1997).

In the Netherlands, however, this method recently has been applied (implicit): a 'convenant' (agreement) between the Dutch government and the Dutch chemical industry, where "benchmarking of the world best practice" is used to agree on the measures to be taken to reduce the several emissions. (Benchmarking is a modern management technique, used in many companies for medium term target setting. One might argue whether this technique generates targets which are 'stretched' enough for sustainability purposes. Note, however, that targets set by benchmarking are 'moving targets', since a 'best practice' is getting better and better over the years, a process caused by competition).

It is evident that a perfect weighting principle does not exist. How to overcome this problem with a different approach is discussed in the next section.

2.4 The development of a new model

Since there is a need for a single indicator (as explained in Chapter 1), and since there appeared to be no satisfactory existing one, the EVR model has been developed, based on the following criteria:

1. The model has to be easily explainable to non-experts (i.e. the model has to relate to "normal life").
2. The model has to be 'transparent' for specialists:
   - Since the choice of the region influences the calculations in the model, specialists have to be able to adapt the data to cope with the choice of a specific region (the first calculations of the model have been made for the Dutch and/or Western European region)
   - Since the character of these calculations is that some arbitrary decisions cannot be avoided, the model has to have a structure that enables an easy assessment of the effect of these decisions, so that the experts can adapt the model to their own judgements.

Based on the analyses of Section 2.3, it was concluded that a model based on the 'marginal prevention costs' (see Appendix 2 and 3) seems to give the best fit with the two criteria mentioned above:

1. The idea of prevention costs is easy to explain to non-experts (everybody is aware of the fact that measures to prevent emissions will cost extra money).
2. The idea of marginal costs is easy to explain to non-experts (what does matter is the most expensive measure our society is prepared to take; strategies of introduction are easy to explain and discuss as well; consequences for business strategies are easy to explain and discuss).
3. The methodology of marginal cost calculations is 'transparent' as such for experts: experts can follow each step of the calculation and judge whether they agree on the data which is used, and they can assess the sensitivity for uncertainties of assumptions.
4. Experts can make calculations for different regions.

The problem, of course, is how to deal with setting the sustainable norms for emissions (see Section 2.3, point 1, 2 and 3). The chosen strategy for this problem is: keep the model as simple as possible (so it remains transparent). This is achieved by the following methodology:

Step 1 Estimate which set of measures (technical solutions, 'end of pipe' and/or process integrated) will meet the requirements for sustainability

Step 2 Relate these arbitrary norms to calculations of 'absolute' norms in literature and relate them to governmental (political) aims

Step 3 When the chosen norms of step 1 is not satisfactory in step 2, reset the norms in step 1 and repeat step 2; when the norm is OK, take the price of the most expensive measure.

In this way, the complex calculation systems on 'absolute' norms (and the scientific discussions about them) are not integrated in the model, but are kept separate from the model on the marginal prevention costs. This separation of models is essential to keep the total system transparent (the complex situation with regard to greenhouse gases is a good example, see the Chapter 3).

2.5 The norms in the model and how they relate to other norms and aims

Applying the methodology of the previous chapter, the following norms are proposed:

1	prevention of acidification	7.55 €/kg SOx equivalent
2	prevention of eutrophication	3.60 €/kg phosphate equivalent
3	prevention of ecotoxicity (heavy metals)	802 €/kg Zn equivalent
4	prevention of carcinogens	33 €/kg PAH equivalent
5	prevention of summer smog	8.90 €/kg C_2H_4 equivalent
6	prevention of fine dust (winter smog)	27.44 €/kg fine dust PM2.5
7	prevention of global warming	0.135 €/kg CO_2 equivalent.

The relationship with other norms for sustainability will be dealt with hereafter.

2.5.1 Global warming

The norm of 0.135 €/kg CO_2 equivalent relates to the lists of prevention measures of Table 2.2, for reduction of greenhouse gases (end of pipe as well as process integrated measures). The list is a summary of measures that are technically feasible at current price levels used in the MARKAL (Beeldman, 1998, Gielen, 1998) and the MATTER (Gielen, 1999) models of ECN (ECN, 1998), Petten. The list applies to the Netherlands, but the list for Western Europe shows only minor differences (Gielen, 1998 and 1999). The importance of such a list is, that it shows which measures are included and which are excluded at a certain price level. It provides the reader with a feeling for the economic feasibility of certain types of measures:

1.	biomass for production of electricity	25 – 60	€ / 1000 kg CO_2 equ
2.	CO_2 storage at production of electricity	60 – 95	€ / 1000 kg CO_2 equ
3.	Renewables (windmills, solar heating systems)	95 – 135	€ / 1000 kg CO_2 equ

At this price level, some measures are excluded, such as biofuel for cars and Photo Electric Cells.

Calculations with the MARKAL and MATTER models for the Netherlands show that, starting a reduction programme in 1999, the Kyoto norm (- 6% in 2010 in relation to the level of 1990-1995) can just be reached at a norm of 95 €/ 1000 kg CO_2 equivalent. Calculations with MATTER show that 135 €/ 1000 kg CO_2 equivalent can result in a reduction of 50% in 2020 (compared with the year 2000) for Western Europe as well as the Netherlands[7].

The aforementioned calculations show that the choice of 135 €/1000 kg CO_2 equivalent is somewhat arbitrary indeed[8]:

1. The Kyoto norm could have been met with 95 € (if immediate actions had been taken).
2. Renewables come only in at a level of 95 – 135 €.
3. Is a reduction of 50% (of level 2000) for Western Europe in 2020 enough? (a factor 4 can be reached at 600 € according calculations in MARKAL and MATTER)[9]

Table 2..2. List of measures for reduction of greenhouse gases in order of rising costs. Western European price level, 1998, updated to price level 2007 (Beeldman, 1998, Ybema, 1995).

	Costs (€ / 1000 kg CO_2 equivalent)	Short description
1	0 (or negative)	1999 tax increase on petrol
2	0 (or negative)	2003 tax increase on petrol
3	0 (or negative)	Greening of taxes on cars (1)
4	0 (or negative)	Less fuel consumption cars 1999
5	0 (or negative)	Less fuel consumption cars 2003
6	0 (or negative)	Energy campaign on cars
7	0 (or negative)	Differentiating tax on new cars
8	0 (or negative)	Differentiating tax on existing cars
9	0 (or negative)	Increase of tire pressure
10	0 (or negative)	Stringent control on current speed max.
11	0 (or negative)	Energy savings of domestic appliances
12	0 (or negative)	Cruise control, etc. in cars
13	0 (or negative)	Max. speed trucks 80 km/hr , stringent control
14	0 (or negative)	Max. speed cars 100 km/hr , stringent control
15	0 (or negative)	Energy savings in domestic houses
16	0 (or negative)	Existing level of nuclear power
17	0 (or negative)	Energy savings in industry
18	0 (or negative)	Energy savings in farms
19	0 (or negative)	Commuting more by car sharing or by public transport
20	0 (or negative)	More public transport for short distances
21	0	PFCs reduction in the aluminium industry

7 These calculations assume that nuclear power is eradicated and is replaced by sustainable energy sources. Note that these calculations are based on technical measures and have nothing to do with the political discussions on the subject (the political choices of targets).

8 With regard to the calculation of the marginal prevention costs of CO_2, it was decided to take the costs of offshore windmill parks (replacing electrical power from coal fired plants) as the most expensive measure of the prevention curve. This norm for sustainability can continuously be checked, since more and more data of feasibility studies become available.

9 Such a factor requires the implementation of hydrogen fuel cells for vehicles (the price of that is not known yet) and electrical cars, as well as introduction of PV cells in the Netherlands (PV cells is not a very cost effective measure in the Netherlands).

22	0.23	HFC reduction by means of afterburners in industry
23	1,9	N_2O reduction in nitric acid production
24	4,9	Biochemical reduction of methane emissions at organic waste
25	5,3	Oxidation of methane at organic waste
26	10.5	Early closure of coal fired power plants
27	10.5	Replacing HFCs for coolants
28	13	Reduction methane emissions at gas fields
29	13	Replacing HFCs in hard foam
30	13	Replacing HFCs for aerosols
31	13	Recycling of HFCs for coolants
32	13	Reduction of HFC emissions in production of “closed” foam
33	13	Reduction SF6 emissions in chips industry
34	13	Reduction SF6 emissions from power switches
35	24	Bio mass for industrial heat and power plants
36	24	Carbon black for heat and power plants
37	24	CO_2 storage (underground) at refineries and ammonia production
38	30 – 130	Emission reduction in agriculture
39	26	Replacement of coal by gas in power plants
40	41	District heating near power plants
41	47	Reduction of HFC leakages
42	59	Reduction of methane emissions by manure processing
43	55	Gasification of biomass
44	65	CO_2 storage at new gas fired power plants
45	71	Energy savings by domestic heat pumps
46	77 – 183	Domestic solar heating (boiler) systems
47	77	Import of bio mass for industrial heat and power plants
48	83	Certificates for industry
49	83	NO_2 reductions from traffic
50	83	CO_2 storage at coal fired power plants
51	94	Nuclear power plants, new (this measure is skipped for obvious reasons)
52	90	CO_2 storage existing gas fired power plants
53	106	Energy savings in existing industrial buildings
54	50 – 150	Wind energy, on shore
55	165	Biofuels for cars
56	180	Expansion Dutch forests
57	210 –230	Domestic ‘low energy’ houses
58	120 – 270	Wind energy, off shore
59	170 – 290	‘Low energy’ buildings (new) in the industry
60	220 – 550	Further expansion Dutch forests
61	600 - 770	Photo Electric cells in the Netherlands

2.5.2 Acidification

The norm of 7.55 €/kg SO_x equivalent for acidification relates to a list of 141 measures of RIVM, Bilthoven. This list of measures comprises all sectors of society (industry, agriculture, buildings and houses, transport, etc.). Economically feasible at the norm are 121 measures on this list, all based on commercially available technologies. Included are (among others):

1. Measures related up to the EURO-3 norms for cars, trucks, busses and tractors.
2. A vast list of measures for low No_x emissions in power plants.
3. 'Low emission stables' and 'equilibrium nutrification' practices for fertilization of land.

IVM, Amsterdam, used the RIVM database to make a calculation for the Dutch situation based on the year 1992 (Dellink, 1997). See Figure 2.3. Applying the norm for the marginal prevention costs of 7,55 €/ kg SO_x equivalent to the curve of Figure 2.3 results in an emission reduction 750 million kg SO_x equivalent per annum.
Calculations of IVM (Dellink, 1997) suggest an emission reduction of 635 million kg So_x equivalent per annum (the calculation ranges from 485 to 775 million kg SO_x equivalent per annum).

To reach the emission norm of the Dutch government of 240 million kg SO_x equivalent per annum for 2010, the emission reduction has to be 720 million SO_x equivalent per annum.

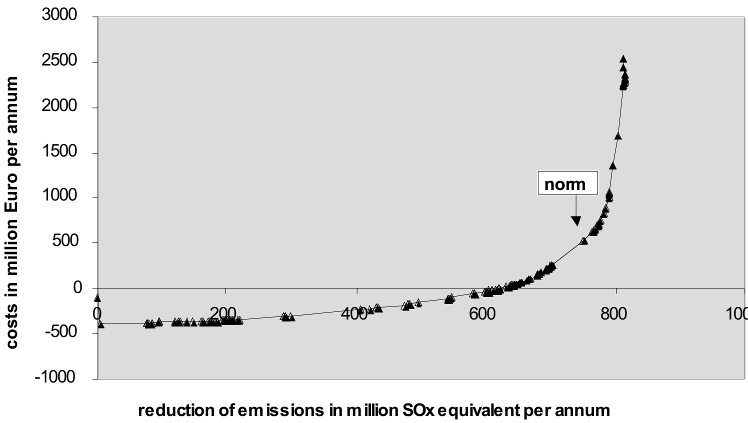


Figure 2.3. The emission reduction curve for acidification in The Netherlands. Source IVM, Amsterdam (Dellink, 1997) original figure.

Note 1. The negative costs at the left end of the curve result from maximum speed restrictions for cars (90 km/hr) and trucks (80 km/hr), resulting in reduced fuel consumption (calculated as savings).
Note 2. The lists of 141 measures don't comprise of high tech solutions such as the introduction of 'green cars', low emission chicken farms, manure conversion techniques, etc. These measures tend to be slightly more expensive than the norm.

Introduction of such techniques, however, will result in a lower slope of the tail-end of the curve.

2.5.3 Eutrophication

Calculations for eutrophication of land for the Netherlands are complex:

1. The pollution within the Netherlands is of the same magnitude as the import and the export by the rivers.
2. The residence time in soil and water is several years, so the "steady state" is complex to assess.

As a result of these factors, calculations and discussions about the subject are rather blurred.

For eutrophication of land a norm has been chosen of 3,60 €/ kg PO_4 equivalent, being the price of sustainable manure processing.

Using the RIVM database for eutrophication, IVM calculated the situation for the Netherlands based on the year 1992. See Figure 2.4. The quantum leap from 15 to 340 million kg PO_4 equivalent per annum is the result of sustainable manure processing.
However, 340 million kg PO_4 equivalent is approximately 50% of the estimated current emission level.
The aim of the Dutch government is a reduction of a factor 4 for the year 2010. This seems to be feasible only when the total production of meat in Holland is reduced drastically, which is already a political discussion in Holland for many years, but which will now come to conclusion under pressure of EC regulations.

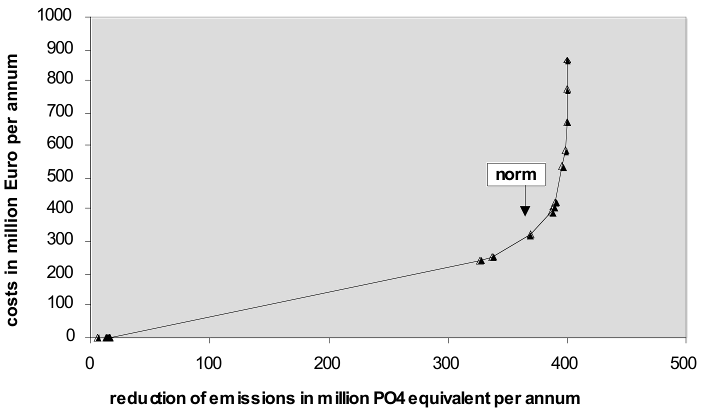


Figure 2.4. The emission reduction curve for eutrophication of land in The Netherlands. Source IVM, Amsterdam, (Dellink, 1997), original figure.

The conclusion is that only a combination of measures (technical process improvements in combination with reduction of production) can lead to a sustainable situation.

2.5.4 Summer smog

The norm of 3.55 €/kg VOC equivalent for summer smog relates to a list of 23 measures of RIVM, Bilthoven. This list of measures comprises measures for industry, energy, building industry, service industry and government and consumers.
The list of measure is supported by an extensive study on the possibilities of reducing VOC emissions in the Netherlands (Cronenberg, 2000).

The original calculation was replaced by a new calculation for the eco-costs 2007. See Figure 2.5. Applying the norm for the marginal prevention costs 3.55 €/ kg VOC equivalent to the curve of Figure 2.5, results in an emission reduction of approx. 180 million kg VOC equivalent per annum.

The norm of 3.55 €/kg VOC is equivalent to 8.90 €/kg C_2H_4 (seeTable 2.1).

Note. A lot of VOC measures are also measures which prevent PAH emissions (like benzene).

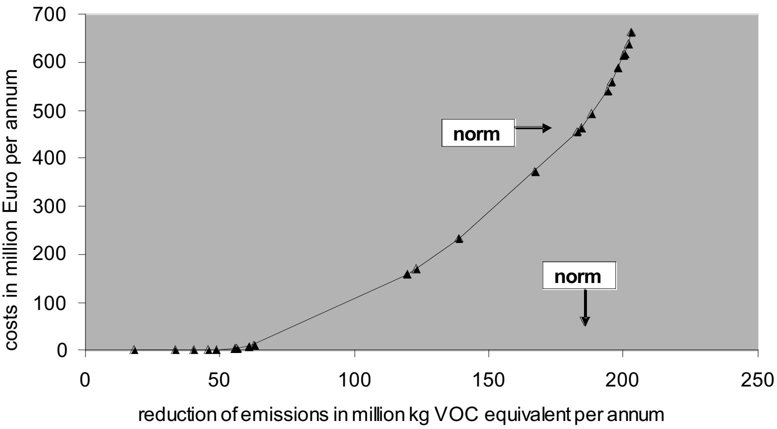


Figure 2.5. The emission reduction curve for VOC (to reduce summer smog) in The Netherlands. Source IVM, Amsterdam, (Dellink, 1997), new figure.

2.5.5 Winter smog

The norm of 14.5 €/kg fine dust PM10 (equivalent to 27.44 €/kg fine dust PM2.5) for winter smog relates to a list of 38 measures for reduction of fine dust of RIVM, Bilthoven. This list of measures is comprised mainly of measures for cars, trucks and busses.
The last measure which determines the marginal costs of 27.44 €/ kg fine dust PM2.5 includes diesel particulate filters for trucks.

IVM, Amsterdam, used this RIVM database to make a calculation for the Dutch situation based on the year 1992. See Figure 2.6. Applying the norm for the marginal prevention costs 14.5 €/kg fine dust PM10 to the curve of Figure 2.6, results in an

emission reduction of approx. 35 million kg fine dust per annum.
Calculations of IVM suggest an emission reduction of 30 million kg fine dust per annum (excluding industry). The calculation ranges from 25 to 35 million kg fine dust per annum.

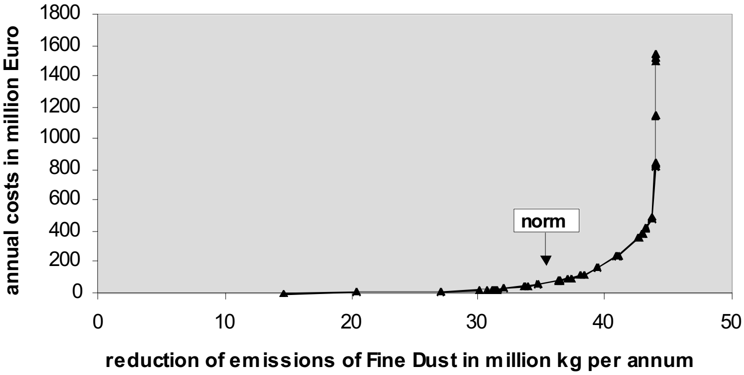


Figure 2.6. The emission reduction curve for fine dust (to reduce winter smog) in the Netherlands. Source IVM, Amsterdam, (Dellink, 1997), original figure.

Note 1. These figures are excluding industry. Industry emissions are 40% of the total emissions and it is assumed that these emissions can be reduced by the same factor for even a lower price.
Note 2. Measures against acidification and global warming effect fine dust as well.

2.5.6 Heavy metals

The norm of 802 €/ kg for heavy metals relates to a list of 14 measures for reduction of Zinc of RIVM, Bilthoven. This list of measures comprises mainly of measures for construction materials.
Zinc has been selected to be the norm for heavy metals, since the emissions of Zinc count for about 60% (weight) of the total heavy-metals emissions.
The last measure which determines the marginal costs of 802 €/kg Zinc is replacement of Zinc by coatings of construction materials (replacement of galvanized steel)

IVM, Amsterdam, used this RIVM database to make a calculation for the Dutch situation based on the year 1992. See Figure 2.7. Applying the norm for the marginal prevention costs 802 €/kg Zinc to the curve of Figure 2.7, results in an emission reduction to water of approx. 250,000 kg Zinc per annum.
Calculations of IVM suggest an emission reduction of 250,000 kg Zinc per annum is required. This calculation is rather inaccurate and ranges from 107,000 to 407,000 kg Zinc per annum.
The emission norms for heavy metals are still under discussion, since the effect of heavy metals is strongly influenced by local conditions.

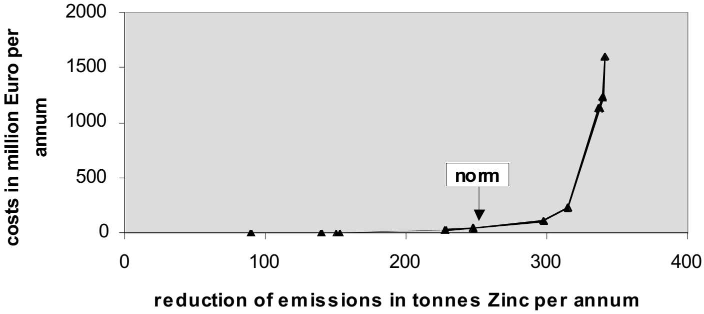


Figure 2.7. The emission reduction curve for Zinc in The Netherlands. Source IVM, Amsterdam, (Dellink, 1997), original figure.

2.5.7 Carcinogens

For carcinogens there is no calculation of the prevention costs curve available. As a norm, 33 €/ kg PAH equivalent for carcinogens has been taken[10].

2.6 Example: the pollution prevention costs '99 of paper

As an example, a calculation of the pollution prevention costs '99 of the production of paper (wood based, chlorine free bleached white printing paper) is given in Table 2.3. Data are from BUWAL (Oekobilanz von Packstoffen, 1990), Bern.

[10] In the eco-costs 99 it was the same norm as the norm for dust (since fine dust particles, in industrial areas often micro droplets of chemicals, plays an important role in cancer). After discussions with Pré Consultants, this has been changed. From the study on prevention of VOC, it appeared that industrial prevention measures of PAH have the same nature as prevention measures on carcinogens. The most expensive prevention measure of this study was taken as a norm for carcinogens: €33 per kg (= the "best practice" available to reduce emissions from storage systems), resulting in a maximum reduction of PAH emissions (a maximum of precaution).

Table 2.3. An example of the calculation of the pollution prevention costs of wood based, chlorine free bleached white printing paper, quantity 1 kg (Data from BUWAL Oekobilanz van Packstoffen, 1990, Bern).

1 pollutant	2 class	3 amount (kg)	4 characterization factor	5 3 x 4 "kg equ."	6 poll. prev. costs (€/kg)	7 5 x 6 poll. prev. costs (€)
ammonia	acidification	3.63E-06	1.88	6.82E-06		
HF	acidification	1.20E-08	1.6	1.92E-08		
NO_x	acidification	5.47E-03	0.7	3.83E-03		
SO_2	acidification	1.22E-02	1	1.22E-02		
			subtotal:	1.60E-02	7.55	0.12
ammonia	eutrophication	3.63E-06	in acidification [11]			
NO_x	eutrophication	5.47E-03	in acidification			
COD	eutrophication	4.46E-02	0.022	9.81E-04		
NH_3	eutrophication	1.03E-06	0.33	3.40E-07		
			subtotal:	9.82E-04	3.60	3.5E-03
CO_2	greenhouse ef.	1.61	1	1.61		
N_2O	greenhouse ef.	3.59E-04	289	0.10		
			subtotal:	1.71	0.135	0.23
Hg in air	heavy metals	1.90E-08	11.26	2.1E-07		
Hg in water	heavy metals	1.00E-09	11.26	1.1E-08		
			subtotal:	3.2E-08	802	2.5E-05
aldehydes	summer smog	1.02E-05	0.657	0.67E-05		
CxHy	summer smog	6.98E-03	0.398	6.98E-03		
			subtotal:	7.0E-03	8.9	6.2E-02
dust (SPM)	winter smog	4.57E-03	0.157	7.2E-04		
SO_2	winter smog	1.22E-02	in acidification			
			subtotal:	7.2E-04	27.4	1.9E-02
		Sum of 7:	Total pollution prevention costs '99, 1 kg paper[12]:			€ 0.43

Note. The LCA of paper can vary considerably with the actual production chain (differences in production plants of pulp and of paper and differences in transport chains). Typical chains can deviate by a factor 2 or more from the average. Thus the LCA methodology as well as the method of pollution prevention costs '99 can only be used in *benchmarking* of production chains (comparing two or more cases). Data such as the data in Table 2.3 can never be considered as "the absolute truth".

[11] Ammonia and NOx are in two classes: adification and eutrification. Since the prevention measure of an emission has to be taken only once, these emissions are only counted in acidification (where they have the highest eco-costs), to avoid 'double counting'.

[12] In this example, the BUWAL data are rather old (from 1990, or even older), so the paper of modern paper mills has considerably less pollution (0,2 – 0,3 €/kg).

2.7 Discussion

2.7.1 'Virtual' costs

It is important to mention that the curves of Figures 2.3 through 2.7 are relating to the present and *not* to the future. These curves describe the present state in virtual terms ("what if we already had taken the measures now"). Table 2.2 is also in current prices, the ECN calculations which are referred to in Section 2.5, however, make extrapolations in future years (applying economic growth scenarios).

All measures are readily available technologies at current price levels. It is important to realize that the tail ends of the curves will get flatter (bend to the right) in future because of two effects:

1. Technological learning curves and economies of scale when a technology gets widely applied in the market.
2. Innovation of the technologies of measures and invention of new measures when big potential markets are expected to develop in the foreseeable future (because of the acceptance of a marginal cost level).

Case studies (Jantzen, 1995) on the history of prices for waste water treatment systems (for phosphor) and exhaust gas systems (desulfurization of exhausts of power plants, exhaust systems for cars) suggest that the technical learning effect (price reduction) is 4% - 10% every year over the period of the first ten years in which these systems were introduced in full scale. The history of low NO_x burners shows that innovation resulted in a price reduction of 15% - 30% every year over a period 6 to 10 years.
The conclusion is that one should avoid calculations which go too far into the future, since technologies and prices for measures have not yet been developed.

2.7.2 Why 'marginal prevention costs' instead of 'total prevention costs'?

An important aspect of the model is, that 'marginal prevention costs' have been chosen as norm, where these marginal prevention costs are defined as the maximum costs of a list of selected measures which are assumed to be sufficient to create a sustainable situation ("if we had taken these measures now, we would expectedly have a sustainable situation").
Figures 2.3 through 2.7 suggest that it is also possible to take the 'total prevention costs' as a norm. However, in doing so, the character of the model will change: the 'total prevention costs' are very sensitive for the choice of the region (with certain characteristics of density of population, regional industrial activity, etc.) at the moment in time (on the road to sustainability, these total costs will change constantly, whereas the marginal costs stay constant when the other parameters do not change). See also Appendix 2.

In the marginal costs model, the calculations on the total prevention effects (Figures 2.3 through 2.7) are *only for validation* of the norms: if similar calculations for the areas of Tokyo or Los Angeles show that the marginal costs norms have to be more stringent, the 'virtual pollution costs' have to be adapted accordingly.

The idea of "the prevention costs of the most expensive measure of the list" relates to the idea of applying the 'best practice' (in terms of technical feasibility and economic optimum) for sustainability. The best practice approach requires that the best practice will be applied in a total region, regardless of the fact that parts of that region could cope with less than the best practice (Example: In the Netherlands there is only a serious summer smog problem in the Rotterdam area, however, national emission norms are applied to the whole country to prevent 'export' of environmental problems to relatively clean sub-regions). It is a political decision (the political will) to which area (the World, the Western World, the European Community or to one country) the norm will be applied. Only when norms are set for the whole World, problems such as 'exporting environmental problems' and 'levelling the commercial playing field' can be resolved definitely.

Note 1. The best practice approach is already accepted within some big multinational companies. (Unilever has an environmental policy that *current* best practice technologies which are applied in e.g. Holland, also have to be implemented in production facilities in other parts of the world. Shell is trying to implement this way of thinking in terms of their Norms and Values as well)
Note 2. It is not allowed to add-up the total costs of prevention of Figures 2.3 through 2.7 to calculate the 'grand total costs of prevention' for the region. The reason is that some measures do have an impact in more than one figure, resulting in 'counting double for one measure'. Note that this effect does not influence the marginal prevention costs model.

2.7.3 How to deal with other prevention costs than of these 7 classes?

There has been a recent tendency to take many more classes into account. See Table 2.4 for some leading models. The model of the virtual eco-costs (where depletion of materials and fossil fuels is also taken into account), which will be introduced in Chapter 3, is an example of that.

We think, however, that the following 3 issues should be dealt with separately:

1. the original sustainability issues (pollution and depletion of the earth), which have a wider impact than just local and temporary effects
2. local health and safety issues (including the local damage of noise and smell, emission levels inside manufacturing facilities, etc.)
3. issues related to the conservation of nature (related to urban planning, planning of national parks, global master planning, etc.)

When these 3 issues are mixed up, the political discussions will get blurred. This is, for instance, debated in The Netherlands as to where to plan the future Amsterdam Schiphol Airport (to build an airport in the North See is a fair proposition from the point of view of health and safety, but not from the point of view of sustainability and/or conservation of nature).

The model which is presented in this chapter is a model for the first category only (point 1.). It is not meant to deal with the other two categories (point 2. and 3.). The model of Chapter 6 goes beyond the use in LCAs only. This model is applicable in urban planning as well.

Table 2.4. Classification in some models which are used in product design[13]

	Eco-costs model Vogtländer	Eco-indicator '95	Eco-indicator '99	EPS	NSAEL	SETAC
Eutrofication	+	+	+		+	+
Acidification	+	+	+		+	+
Ozone layer		+	+		+	+
Carcinogens	+	+	+			
Global warming	+	+	+		+	+
Heavy metals	+	+	+			
Winter smog	+	+	+			
Summer smog	+	+	+			+
Pesticides		+				
Noise	(+)					+
Smell						+
Radiation						+
Health	+		+	+		
Human toxicity	+		+		+	+
Respiration	+		+			
Depletion materials	+		+	+		
Loss of agric. production	+			+		
Damage ecosystems	+		+			
Ecotoxicity	+		+			+
Biodiversity	+			+		
Scenic beauty	(+)					
Casualties						+
Aesthetic values				+		
Econ. value	+					

[13] References: Eco-indicator '95: (Goedkoop, 1995), Eco-indicator '98: (Goedkoop, 1998), EPS: (Steen, 1996), NSEAL: (Kortman, 1994), SETAC: www.ecomed.de

2.8 Call for Comments

1. We are very interested in similar calculations for other regions outside the Netherlands:
   - Do you have similar calculations (referring to Figures 2.3 – 2.7) for any of these classes or for other pollution classes?
   - For CO_2 reduction: what are the costs for CO_2 reduction measures in your region and at what level of marginal prevention costs does your region comply with 'Kyoto', and at what level do you expect a 50% reduction?
   - Do you have any examples of 'industrial best practices' and benchmarking, and what are the norms in these examples (emission levels, emission prevention levels, emission prevention costs)?
2. Do you have any suggestion to extend the classes with another class, and how do you arrive then at the marginal prevention costs for that class (e.g. hindrance of noise)?
3. Especially for developing countries, it is possible to make a quick estimate of the pollution prevention costs (1. assess the regional environmental problems; 2. make a list of measures to be taken; 3. determine the marginal prevention costs for each class).

   This could result in a set of data for each different regions. Such a calculation model however makes sense only when the Life Cycle Inventory of emissions does take into account the region where the emission occurs, which adds quite some complications to the current LCA methodology.

   Do you feel there is a need for such an enhancement of the LCA methodology? Why and for which type of situations? Or do you feel that the LCA methodology should be kept simple?
4. The underlying idea of point 3 is that the developing countries cannot afford the prevention measures of the western world, and they don't need them (because their emission levels are low).
5. However, one may argue differently: in order to gain maximum environmental protection, best practices in the field of prevention measures should be applied world wide and 'export of environmental problems for economic reasons' should be suppressed.

   Such an approach would require world wide standards for prevention measures and/or prevention costs (in € or US $ per kg equivalent per class). In such a model regions with high emissions will have a high economic burden to prevent these emissions, regardless of there own sustainability norms and there economic situation. As a consequence the western world has to subsidize the developing countries where necessary.

   How should we arrive at such world wide norms? Do we expect then norms which will be totally different from the norms presented, and if so why? When you have comments on these questions and/or you have comments on any specific aspects of the calculation method which has been presented here, please mail the corresponding author[14].

[14] 8 out of 9 of the people that responded were in favour of a global set of data, following the idea that "best practices" should be adhered to at a world wide level. Nevertheless, universities in Japan, South Korea and China did make calculations on marginal prevention costs for their own country/region.
Particularly a group of Universities in Japan has been successful in developing their own set of data (Y. Fujii, Bunkyo University; T. Oka, Fukui Prefectural University; M. Ishikawa, Tokyo University of Fisheries; S. Susami, Ebara Corporation; H. Matsuno, Meiji University; S. Kato, Tokyo University), under the name of 'maximum abatement costs'.
In the Netherlands, the owners of the "GreenCalc" computer software desided in 2002 to adapt als the marginal prevention costs as a basis for their LCA calculations. The Eindhoven University of Technology made new calculations (which differed not considerably from the original dataset of the eco-costs 99). Greencalc, however did not yet update their tables to the eco-costs 2007 standard. Note that the reasons to introduce the new dataset of eco-costs (the eco-costs 2007) have been:
• The availability of better characterization tables (CLM-2, Impact 2002+, IPPC 2007)
• The availability of new studies on prevention costs (of VOC, PAH and CO_2)

3 The Eco-costs and the EVR[15]

3.1 Abstract

This chapter deals with the development of a single indicator for eco-efficiency.
It also decribes an enhancement of the allocation methods in the current LCA methodology, to cope with the allocation problems in calculations of services

In literature many models (qualitatively as well as quantitatively) can be found to cope with the problem of communicating results of LCA analyses with decision takers. In the previous chapter, an LCA-based single indicator for emissions is proposed: the virtual pollution prevention costs '99.

In this chapter, a single LCA-based indicator for sustainability is proposed. It builds on the virtual pollution prevention costs '99 for emissions, and adds the other two main aspects of sustainability: materials depletion and energy consumption. This single indicator, the 'virtual eco-costs 99', is the sum of the marginal prevention costs of:

- Materials depletion, applying 'eco-costs of materials depletion', to be reduced by recycling
- Energy consumption, applying 'eco-costs of energy', being the extra price of renewable energy
- Toxic emissions, applying the virtual pollution prevention costs '99.

The calculation model includes 'direct' as well as 'indirect' environmental impacts. The main groups of 'indirect' components in the life cycle of products and services are:

- Labour (the environmental impacts of office heating, lighting, computers, commuting, etc.).
- Production assets (equipment, buildings, transport vehicles, etc.).

To overcome allocation problems of the indirect components of complex product-service systems, a methodology of economic allocation has been developed, based on the Eco-costs / Value Ratio (EVR).
This EVR calculation model appears to be a practical and powerful tool to assess the sustainability of a product, a service, or a product-service combination.

[15] Original title: "The Virtual Eco-costs '99, a single LCA-based indicator for sustainability and the Eco-costs/Value Ratio (EVR) model for economic allocation." Published in Int. J. of LCA (Vogtländer, Brezet, Hendriks, 2001,B).

3.2 Introduction: the philosophy behind the model

In March 1995, the World Council for Sustainable Development defined eco-efficiency as:

> "the delivery of competitively priced goods and services that satisfy human needs and bring quality of life, while progressively reducing ecological impacts and resource intensity, throughout the life cycle, to a level at least in line with the earth's estimated carrying capacity." *(WBCSD, 1995)*

This business oriented definition links modern management practice ("the delivery of competitively priced goods and services (…) quality of life") to the need of a sustainable society ("while progressively reducing … to … earth's carrying capacity"). The first part of the sentence asks for a maximum value/costs ratio of the business chain, the second part of the sentence requires that this is achieved at a minimum level of ecological impact.

But what does this rather philosophical definition mean to business managers, designers and engineers in terms of the practical decisions they take?

There is a need to resolve simple questions such as: what is the best product design in terms of ecological impact?, what is the best product portfolio in terms of sustainability?, what is the best sustainable strategy?.

For that reason, the Delft University of Technology developed the eco-costs / value model as a practical tool for decision-making, based on the LCA methodology, and comprising the following features:

- one single indicator for the 3 major groups of environmental impacts (materials depletion, fossil energy consumption, toxic emissions)
- a relatively simple and well defined allocation model to cope with "service" type functions (as service systems are characterized by many 'indirect' environmental impacts, shared by many other external systems).

The basic idea of the model is to link the 'value chain' (Porter, 1985) to the ecological 'product chain'. In the value chain, the added value (in terms of money) and the added costs are determined for each step of the product 'from cradle to grave'. Similarly, the ecological impacts of each step in the product chain are expressed in terms of money as well: the so-called eco-costs. See Figure 3.1.

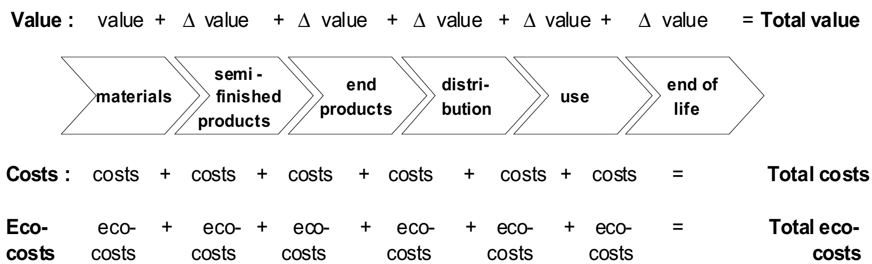


Figure 3.1. The basic idea of combining the economic and ecological chain: 'the EVR chain'.

The eco-costs are 'virtual' costs: these costs are related to measures which have to be taken to make (and recycle) a product "in line with the earth's estimated carrying capacity". These eco-costs are the sum of the 'marginal prevention costs' of each 'class' (type) of pollution[16] (Section 3.4.1) and the costs of measures for prevention of material and energy depletion, see Sections 3.4.2 and 3.4.3.
Since our society is yet far from sustainable, the eco-costs are 'virtual': they have been estimated on a 'what if' basis. The costs of the required prevention measures are not yet fully integrated in the current costs of the product chain (the current Life Cycle Costs). It is expected that, in future, the eco-costs will become part of the product costs (by means of 'eco-tax', 'tradable emission rights', or other governmental measures), since our society will not continue to accept consequences unsustainable situations in the long term.

3.3 The value, costs and eco-costs of a product

Now we look into one step of the business chain.
The value ('fair price') of a product is determined by:

- Product quality.
- Service quality.
- Image.

These 3 components of value are described in more detail by the 'eight dimensions' of Garvin (see Section 5.3).

The cost-structure of a product comprises:

- The purchased materials (or components).
- The required energy.
- Depreciation (of equipment, buildings, etc.).
- Labour.

For each company in the business chain, the tax + profit equal the value minus the costs.
The direct eco-costs have been defined as follows:

- Virtual pollution prevention costs, being the costs required to reduce the emissions in the product chain (from cradle to grave) to a sustainable level.
- Eco-costs of energy, being the price for sustainable energy sources.

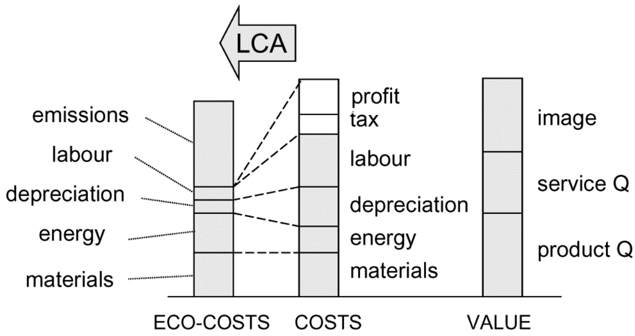
[16] Note that the marginal prevention costs have been chosen here as the norm to be able to compare different kinds ('classes') of sustainability issues. For the logic of such a choice and its implications, see Chapter 2. Basically, the marginal prevention costs are the costs of the last and most expensive measures that have to be taken to bring the economy in a given region to a sustainable level. Marginal prevention costs are not equal to the so-called 'external costs', since 'external costs' are related to damage and not prevention.

- Eco-costs of materials depletion, being (costs of raw materials) × (1 – α), where α is the recycled fraction of materials to make a product (for details on the 'End of Life' and recycling phase, see Chapter 4).

The indirect eco-costs are:
- Eco-costs of depreciation, being the eco-costs related to the use of equipment, buildings, etc.
- Eco-costs of labour, being the eco-costs related to commuting and the use of the office (building, heating, lighting, electricity for computers, paper, office products, etc.).

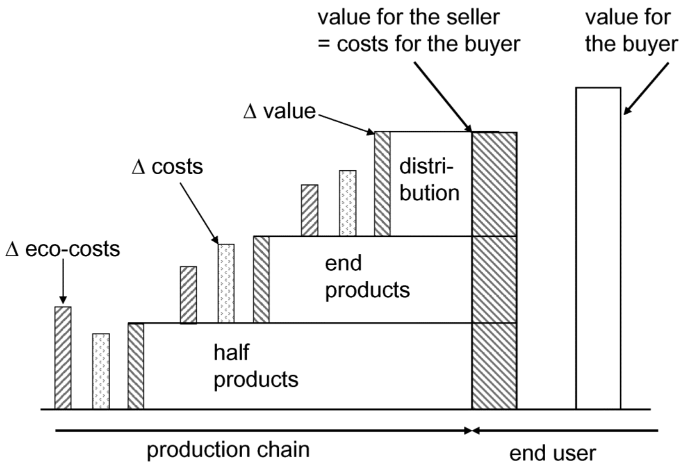
This is depicted in Figure 3.2.

Figure 3.2. The decomposition of 'virtual eco-costs', costs and value of a product.



Along the business chain the value, the costs and the eco-costs can be added up, as depicted in Figure 3.3.

Figure 3.3. The decomposition of the value in the chain.



Characteristic for each process, product or service is the ratio of the value and the eco-costs.

We can define this Eco-costs/Value Ratio, EVR, at every aggregation level of the chain (or pool[17]): EVR = eco-costs/value.
A low EVR indicates that the product is fit for use in a future sustainable society. A high EVR indicates that the value/costs ratio of a product might become 'less than one' in future (since 'external' costs will become part of the 'internal' cost-structure), so there is no market for such a product in future.
Later in this chapter we can see how we might apply the EVR model for economic allocation in the LCA of a complex product-service system, but first we will define and describe the eco-costs.

3.4 The components of the eco-costs

As mentioned in Section 3.2, we define the eco-costs as the sum of 3 direct (a, b and c) and 2 indirect (d and e) elements:
a) the virtual pollution prevention costs
b) the eco-costs of energy
c) the eco-costs of material depletion
d) the eco-costs of depreciation (use) of equipment, buildings, etc.
e) the eco-costs of labour.

All these elements are calculated according to the LCA method, as defined in ISO 14041, as described hereafter.

3.4.1 The virtual pollution prevention costs

The virtual pollution prevention costs have already been introduced in the previous chapter.
Table 3.1 provides the data for a list of 29 materials. Note that the data in this Table also include the pollution prevention costs related to the emissions of the use of energy to produce and transport these materials. For an extensive list of over 5000 materials and processes, see www.ecocostsvalue.com.

[17] (Gadiesh, 1998), see also Section 3.5

Table 3.1. A summary of the eco-costs 2007 of emissions of materials production; based on LCA's from ecoinvent v2 and Idemat 2008.

MATERIALS	VIRTUAL POLLUTION PREVENTION COSTS 2007 = eco-costs 2007 for emissions (€/kg)
Aluminium, virgin	2,22
Secondary aluminium	0.27
Copper, virgin	2,06
Secondary copper	0.22
Stainless steel (market mix: 40% recycled, 60% virgin)	1,44
Steel (market mix)	0.31
Steel (100% recycled)	0.12
Glass	0.15
Glass wool	0.30
Ceramics	0.04
Paper board	0.12 - 0.22
Paper	0.31
Wood	Various data, see www.ecocostsvalue.com
Recycled paper	0.08
Concrete (excluding reinforcement)	0.025
Acylonitril-butadiene-styrene	0.63
High density polyethylene	0.32
High impact polystyrene	0.57
Low density polyethylene	0.35
Polyamide	1,51
Polyethylene tereftalat (PET)	0.57
Polypropylene	0.32
PPE/PS	0.56
Polystyrene hard foam	0.54
PUR	1,0
PVC	0.33
Rubber (natural)	0.19
Stone wool	0.38

3.4.2 The eco-costs of energy

The calculation method to determine the eco-costs of energy is based on the assumption that fossil fuels have to be replaced by sustainable energy sources. The 'eco-costs of energy' is equal to the extra costs of the renewable energy system which has to replace the current system. The data which is used, is from the MARKAL database of ECN (ECN, 1998, Gielen, 1998). See also Table 2.2. The results of the calculations for 6 sources of energy are given in Table 3.2. The technologies are selected from MARKAL. These technologies are readily available. It might be, however, that the costs of the proposed domestic sustainable energy systems become gradually lower when the techniques will be widely spread (because of the economies of scale of production costs for a big consumer market).

For an extensive list, see www.ecocostsvalue.com.

Category	Main sustainable energy source	Eco-costs of energy 2007 (€/GJ)
Industrial heat	Biomass	11.82
Diesel (including combustion)	Ethanol from biomass	29.87
Electricity (industry)	Biomass +wind	26.27
Electricity (domestic)	Biomass +wind	30.20
Petrol (including combustion)	Ethanol from biomass	29.87
Domestic heating	Suncollectors+heatpumps	13.5

Table 3.2. The eco-costs 2007 of energy.

3.4.3 Eco-costs of materials depletion

With regard to the depletion of materials, the main approach in the model is:

- The eco-costs of materials depletion are set equal to the market value of the raw materials when the materials are not recycled.
- When a fraction α of the sourced material is recycled, a factor $(1 - \alpha)$ is applied to the market value of the raw material for the new product to calculate the eco-costs of materials (for the general concept and its details see Section 3.4).

Therefore:

(3.1) eco-costs of materials depletion = 'market value of the raw material' $\times (1 - \alpha)$

The underlying assumption is that the price of the virgin material for metals has to increase 100% to reach a sustainable level..[18]

For plastics however, the situation is different, since the source of it is crude oil.
The average crude oil price was $ 15,50 per barrel for the period 1994-1998, but became very volatile in recent years. This price level is also valid for much longer periods in history (for the long term average crude oil prices see www.wtrg.com.), but nobody can predict prices in the future: is $ 60.00 - $ 100.00 per barrel realistic?
However, it is more in line with the general philosophy of this model to avoid the use of fossil fuels and use biomass instead as source material for plastics.
Therefore, in the EVR model the price of feedstock for plastics based on biomass has been chosen for the eco-costs of materials depletion. This price is estimated at 0.7 €/kg.

[18] For the short term, one has to apply here the 'present market value' (discounted) of the 'sustainable alternative in the future' for the metal which is depleted, according to the model of Hotelling (Pearce, 1990)(Henley, 1997). This cost based approach, however, is not valid in the last decade because of heavy financial speculations. So calculations have been based on price levels of 1990 - 1999. An increase of 100% of those price levels is expected as 'long term sustainable price level', based on a sustainable demand in the period 2050 - 2100, which is estimated at 3 times the current demand. This price increase is equal to the eco-costs. For details on the calculation and assumptions see www.ecocostsvalue.com.

The fractiona α has to be applied to the materials used for the new product (and not as a fraction of materials from the old product at the End of Life), after the upgrading process (if applicable). See Figure 4.5.

Table 3.3 provides the material depletion costs for 46 metals, 15 chemicals, and 44 plastics. (The basis for the data on metals is the average market values for the period 1988 – 1998, with trend were applicable. The basis of the chemicals and plastics is the amount of oil 'stored' in the product)

Table 3.3. The eco-costs 2007 of materials depletion; derived from: Metal prices in the USA, Mineral Information Team, USGS, applying linear regression for the period 1988 – 1998 and the Production Price Index for 1998-2007.

Materials		Ecocosts 2007 (€/kg)	Materials	Ecocosts 2007 (€/kg)
Metals (in ore)			Chemicals	
Aluminium	Al	1.65	Acrylonitril	0.125
Antimony	Sb	3.19	Acetic acid	0.327
Arsenic	As	1.18	Acetic anhydride	0.384
Beryllium	Be	826	Benzene	0.639
Bismuth	Bi	9.44	Bisphenol A	0.645
Cadnium	Cd	2.36	Epichlorohydrin	0.318
Calcium	Ca	5.31	Formaldehyde	0.327
Cesium	Cs	1.77	MDI	0.280
Chromium	Cr	9.44	MMA monomer	0.280
Cobalt	Co	53.1	Pentane blowing agent	0.280
Columbium (Niobium)	Nb	8.26	Phenol	0.280
Copper	Cu	2.24	Polyether-polyols	0.280
Gallium	Ga	472	Propylene	0.700
Germanium	Ge	1416	Styrene	0.754
Gold	Au	11800	TDI	0.493
Hafnium	Hf	236		
Indium	In	354	Plastics thermoplast	
Iron	Fe	0.12	Plastics (general)	0.700
Lead	Pb	0.94	ABS 30% glass fibre	0.488
Lithium	Li	94.4	ABS	0.697
Magnesium	Mg	4.13	PA 6 GF30	0.385
Manganese	Mn	0.0059	PA 6	0.550
Mercury	Hg	7.08	PA 66 GF30	0.385
Molybdenum	Mo	5.9	PA 66	0.550
Nickel	Ni	8.26	PB	0.754
Platinum	Pt	15104	PC 30% glass fibre	0.458
Palladium	Pd	5664	PC	0.655
Iridium	Ir	7080	PE (HDPE)	0.700
Osmium	Os	15340	PE (LDPE)	0.700
Rhodium	Rh	18880	PE (LLDPE)	0.700
Ruthenium	Ru	1770	PE expanded	0.700
Rhenium	Re	1180	PET 30% glass fibre	0.357

Rubidium	Rb	885	PET amorph	0.510
Selenium	Se	10.62	PET bottle grade	0.510
Silicon	Si	0.17	PMMA	0.490
Silver	Ag	188.8	PP GF30	0.490
Tantalum	Ta	82.6	PP	0.700
Tellurium	Te	59	PS (EPS)	0.754
Thallium	Tl	23.6	PS (GPPS)	0.754
Thorium	Th	47.2	PS (HIPS)	0.754
Tin	Sn	10.62	PVC (b)	0.314
Titanium	Ti	10.62	PVC €	0.314
Tungsten	W	0.0708	PVC (s)	0.314
Vanadium	V	25.96	PVC	0.314
Zinc	Zn	1.42	PVDC	0.202
Zirconium	Zr	25.96	SAN	0.754
Aluminium (in al. scrap)		0	*Plastics thermosetting*	
Copper (in copper scrap)		0	Epoxy resin	0.622
Steel (in steel scrap)		0	Melamine	0.233
Waste paper		0	MF(resin)	0.280
			PF (resin)	0.350
Bauxite		0.34	Polyester (unsat)	0.700
Iron ore		0.06	PUR flex. block foam	0.598
Manganese ore		0.003	PUR flex. moulded MDI/TDI	0.598
			PUR flex. moulded TDI	0.598
Materials			PUR flex. moulded. MDI	0.598
Oil feedstock		0.7	SMC 25% GL	0.260
Natural gas feedstock		0.7	SMC 50% GL	0.218
Natural gas		0.58 (€/m3)	UF(resin)	0.350
Diesel		0.7		
Petrol		0.7	*Rubber*	
			BR	0.516
			NBR	0.520
			SBR	0.520

3.4.4 Indirect eco-cost: the eco-costs of labour

The eco-costs of labour are indirect eco-costs, since labour as such is hardly causing any environmental burden. However, there is some environmental burden related to labour, such as the environmental impacts of heating, lighting, computers, commuting, etc.

The calculations of these eco-costs are specific for the type of labour. An example is given here for work in offices.

For the category of personnel in offices an assessment has been made for Dutch employees with average salary costs of € 32,500.– per annum (including taxes, insurance and pension funds), having an office space of 33 m^2 (average for the banking and insurance sector):

1. eco-costs of energy per annum per employee (for eco-costs of energy see Table 3.2):
   - commuting by car, 30 km for 210 days per year, fuel required:
   - 1000 litres of petrol = 35 GJ > eco-costs = 35 GJ × 29.9 €/GJ = € 1,046.-
   - heating of the office per annum per employee CBS data[19] for 1994 (CBS 1996)
   - 0.42 GJ/m^2 × 33 m^2 = 14 GJ > eco-costs = 14 GJ × 13.5 €/GJ = € 189.-
   - electricity for the office per annum per employee (CBS 1996)
   - 0.85 GJ/m^2 × 33 m^2 = 28 GJ > eco-costs = 28 GJ × 30.2 €/GJ = € 845.-
2. eco-costs of the office building per employee per annum:
   total costs related to construction, maintenance and demolition
   for details see Table 3.4. > eco-costs = 16.54 €/m^2 × 33 m^2 = € 546.-
3. eco-costs of office products per employee per annum:typical total eco-costs for office products (paper, printing ink, etc.),
   incl. EoL € 200.-

Total eco-costs labour € 2,826.-

Since the average salary costs in this building is € 32,500.–, the EVR ratio is here 0.09. Preliminary calculations show that the eco-costs will rise linear with the salary. So a good guess for labour in offices is EVR = 0.09. Calculations on labour outside offices (shop floor personnel in factories, sales people, truck drivers, etc.) show that the EVR will vary in a range of 0.05 – 0.15, where the eco-costs of commuting and use of electricity play a rather dominant role. Therefore it is recommended to make an LCA assessment in each typical case.

3.4.5 Indirect eco-costs: the eco-costs of depreciation of production facilities

The eco-costs related to the fact that fixed assets are used to make a product, are called 'indirect' eco-costs.

The calculations on the eco-costs of the use of fixed assets have the same characteristics as cost estimates for investments: for each individual situation, calculations have to be made on the applied materials and the required manpower.

The basic idea behind the 'eco-costs of depreciation' is that the eco-costs of the production facilities have to be allocated to the products which are made in or with these facilities.

The standard procedure is:

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[19] The source of the data is the Economic Statistical Institute for the Netherlands, CBS, Voorburg, The Netherlands.

1. Calculate the eco-costs of the production facility.
2. Divide the eco-cost of step 1 by the lifetime T (in years) of the production facility. In the EVR model the 'economic lifetime' has to be applied instead of the 'technical lifetime', which is 'safe side' since the economic lifetime is usually shorter than the technical lifetime.
3. Divide the outcome of step 2 by the number of products N which are produced per year.

In formula:

(3.2) eco-costs of depreciation = (eco-costs of the production facility) / ($T \times N$)

This allocation procedure is similar to the procedure which has been used in the previous section for the calculation of the eco-costs of the office building per employee per annum. Note that different facilities or different subsystems of facilities might have a different lifetime, T (see Table 3.4).

Since the EVR model applies the economic lifetime of the facility, equation (3.2) has a high similarity with the normal, linear equation for production costs related to depreciation:

(3.3) costs of depreciation = (value of the production facility) / ($T \times N$)

Combining equation (3.2) and (3.3):

(3.4) eco-costs of depreciation = (costs of depreciation) × (eco-costs / value)$_{\text{production facility}}$

The meaning for equation (3.4) is that the eco-costs of depreciation can be derived from the normal costs of depreciation by multiplying it with the EVR of the production facility. In situations where more than one type of product is produced in a complex production system, and where a 'cost break down structure' of the product is available, equation (3.4) can provide an easy way out of a rather complex allocation problem.
In Section 3.4 we will show how equation (3.4) can be derived starting from the definition of allocation as stated in ISO 14041.

Calculations show the following characteristics for the EVR:

- Complex machines 0.3
- Luxurious buildings (offices) 0.25
- Low cost offices 0.3
- Processes in stainless steel 0.4
- Refineries 0.5
- Steel structures, cladding 0.5
- Warehouses 0.4

In general:

- The use of steel (and other metals) is related to a high EVR (because of high pollution prevention costs, see Table 3.1).
- Complex systems have a low EVR (because of a high labour content).

Table 3.4. Summary of the eco-costs of an office building (excluding energy during the use phase !), example. For data on eco-costs per kg see www.ecocostsvalue.com. For detailed calculations on houses and offices see www.winket.nl.

Summary description of an office building (typical)	Eco-costs (€/m^2)	Life time (years)	Eco-costs (€/m^2/annum)
Main materials for construction:			
- concrete, 400 kg/ m2 (eco-costs 0,025 €/kg)	10.0	40	0.25
- steel, 50 kg/m2 (eco-costs 0.487 €/kg)	24.5	40	0.61
- miscellaneous materials, 70 kg/m2 (glas, wood, PVC, etc.)	50.0	40	1.25
- construction activities (energy, etc.)	100.0	40	2.50
Subtotal construction building structure	184	40	4.61
Building systems (elevators, heating, electrical, water, etc.)	30	20	1.50
Interior (painting, decorating, furniture,etc.)	75	15	5.00
Computer system (one screen per employee at 33 m2)	6	3	2.00
Maintenance of building and building systems per year	2	1	2.00
Subtotal equipment, interior and maintenance	113		10.50
End of Life:			
Demolition + transport of materials at End of Life	10	40	0.25
Disposal of construction waste (eco-costs 0.118 €/kg)	47	40	1.18
Subtotal End of Life	57		1.43
Total			16.54

3.5 The EVR for economic allocation in the LCA

The reader may already have got a feel of how to apply the EVR model for economic allocation, and how to derive the "total EVR" of a complex system from the EVRs of the system components.
In this section, the EVR is explained in more detail.

An important characteristic of the Eco-costs/Value Ratio of a chain is that the "total EVR" for a production chain is the *weighted average (on value)* of the EVRs of the steps in that chain. This characteristic is shown in the following equation:

(3.5) $$\text{EVR}_{\text{Total}} = \{\sum \text{eco-costs}_n\} \,/\, \text{value}_{\text{Total}} = \sum \{\text{EVR}_n * [\Delta\, \text{value}_n \,/\, \text{value}_{\text{Total}}]\}$$

where:

(3.6) $$\text{value}_{\text{Total}} = \sum\text{costs} + \sum\text{taxes} + \sum\text{profits}$$ (see Figures 3.1, 3.2 and 3.3)

It is obvious that the chain (in its one dimensional form) is a drastic simplification of the real world: in reality production chains are part of production and distribution *networks*. Every actor in the chain is also part of other chains (they have many suppliers and many clients)[20]. In calculation of LCAs this causes the so-called allocation problem: how to allocate the environmental impact of shared use of production facilities, transport and distribution systems, etc.?

The basic methodology for allocation problems in LCAs is dealt with in ISO 14040 and 14044:

> "Where physical relationship cannot be established or used as the basis for allocation, the inputs should be allocated between the products and the functions in a way which reflects other relationships between them."
>
> "For example, environmental input and output data might be allocated between co-products in proportion to the economic value of the products"

This methodology can be explained by an example: the indirect environmental impact of building an air plane, allocated to a single trip[21]. The main parameters are:

- the value of a ticket for the single trip, W, of which a part of that value, X, is related to the depreciation (or leasing costs) of the plane
- the value of a plane, Y
- the eco-costs of a plane, Z (calculated from LCA data).

The question is now which part of the indirect environmental impact of building a plane, Z, has to be allocated to the trip. Applying economic allocation:

(3.7) $EI = (X / Y) \times Z$ = 'the economic proportion' × 'Environmental Impact'

where EI is the indirect environmental impact allocated to the ticket, which can be written as:

(3.8) $\mathrm{EI} = (Z/Y) \times X$ = EVR × 'part of the value of the ticket related to the depreciation of the plane'

Equation (3.8) shows how the EVR model can be used for economic allocation in a complex LCA, starting with a 'cost-breakdown structure'. Especially in cases when proportions of weight are not known directly, which is often the case for services, the

---

[20] This leads to the concept of the 'profit pool' (Gadiesh, 1998)

[21] There is no simple physical relationship to base the allocation on for many reasons. The major two reasons are:

- Planes transport passengers as well as freight (in the same plane on the same trip). How to allocate (split) between passengers and freight? Based on volume or on weight or any combination of both?
- One plane will make many trips during its lifetime, all over the world. There are trips ('legs') with high occupancy rates and trips with low occupancy rates. How to cope with these differences during the lifetime?

EVR model is a powerful tool. Note that equation (3.4) and equation (3.8) are of the same nature.

In the example, equation (3.8) is applied to an 'indirect' environmental impact. Equation (3.8) can also be applied to situations of 'direct' impact (e.g. for allocation of the fuel to one passenger). In most of the situations of 'direct' impact, however, the physical relationship is known as well, in which cases the eco-costs have to be determined on that direct physical relationship, according to ISO 14044.
Although the authors of the ISO 14044 define economic allocation as a 'last option' (to be avoided, if possible) there is no need to avoid economic allocation in cases *where the ratio between 'value' and 'kilograms' is fixed* [22], since the ratio between eco-costs and value, the EVR, is fixed then as well.
So it is a prerequisite for EVR calculations that a specific EVR has to be independent of the size (weight, volume, time, etc.) of the functional unit of the element in the LCA. Under this condition, the EVR can be used for direct impacts as well, instead of the eco-costs / weight ratio, which appears extremely practical in many cases.

The first example is on how to apply the EVR in the case of a service function (a transport chain), where economic allocation plays a major role. This example is given in Table 3.5. In the example of the transport function of Table 3.5, only the 'one way' packaging can directly be linked to the LCA. All other elements in the chain share with other chains (even fuel, since the truck is normally only partly loaded by other freight on the trip back). It is feasible here to establish all 'physical relationships', however, the relationships are of an extremely complex nature, so a computer program has been written to calculate the eco-costs. With the same program structure, costs are being calculated as well.
Analysing the output, it has been concluded that all the activities can be grouped in 'subsystems', since they have the same, constant, EVR (see Section 3.8).
Table 3.5 shows that the calculation of the total eco-costs for the defined function becomes extremely simple, when the values (prices) for the main activities are known, applying these EVR data. The outcome is within 10% of the outcome of the computer calculation based on the 'physical relationships'.

[22] Under such conditions, the 'economic proportion' in equation (3.7) equals the 'physical proportion'

Chain element	LCA subsystem	Value (€)	EVR	Eco-costs (€)
Packaging	(one way boxes)	61	0.16	9.8
Transport	Truck, fuel, road	23	0.58	13.3
Distribution&feeding	Truck, fuel, road	10	0.49	4.9
Storage	Building, forklift truck	6	0.29	1.7
End-of-life	(packaging)	0	0	0.0
Total chain		100		29.7

Table 3.5. An example of using the EVR model for economic allocation in a transport chain. The functional unit is defined as: "transport of 1 litre net volume of tomatoes from Holland to Frankfurt"

The second example is on how to apply the EVR model in the design stage of a product, in this case a warehouse, see Tables 3.6 and 3.7.
In Table 3.6 the classical LCA is provided on the basis of materials required to build the warehouse. This methodology is suited for the situation that the detailed design of the warehouse is finalized (in the Netherlands, two computer calculation models are available for such an analysis in the building industry: Ecoquantum and Kubus Kalk).
However, in the preliminary design stages, the exact amount of materials is not yet known. In that stage the EVR method is more applicable to analyse different alternatives for the design, since the designer has to try to fulfil the design requirements by applying elements with the lowest possible EVR values.
Kubus Kalk is extremely powerfull, since it integrates costs calculation with LCA. There are special databases available in KubusCalc to support the decision taking process during all the design stages of the architect (the Reference Project Method). See for examples www.winket.nl (Dutch).

Table 3.6 and 3.7 provide an analysis on the same warehouse design. The results, however, in terms of eco-costs are not the same: eco-costs of the *activities at the construction site* are often neglected in the case of the classical LCA approach. The preparation of the site, all activities until the floor level is ready, welding, etc., are often underestimated. Looking at the materials only is not sufficient for such an analyses, since the construction phase generates a major part of the ecoburden. See also Table 8.8 with eco-costs of production of parts and eco-costs of construction.

Table 3.6. Eco-costs 2007 for a warehouse, 920 pallets (900 m2, 10 meters high) materials only; calculation according to the classical approach of the LCA method, excl. use phase and End of Life phase.

warehouse 920 pallets, 900 m2, 10 m high	greenh kg CO2 equ	acidific kg SO4 equ	eutroph. kg PO4 eq	photo oxi kg C2H4 equ	fine dust kg PM2,5 eq	aqu ecotox kg Zn eq	mat. depl. (€)
Concrete, reinforced,660000kg	128838	415.4	23.64	25.85	38.32	0.38	7891
Fe360, 51000kg	82832	300.6	42.26	45.43	50.70	0.36	9214
steel sheet, 22000kg	35731	129.7	18.23	19.60	21.87	0.16	3975
PS, 40kg	140	0.4	0.01	0.01	0.01	0.00	30
PS foaming, 40kg	21	0.4	0.00	0.02	0.01	0.00	0.00
steel transforming, 22000kg	6670	19.2	6.52	0.42	0.91	0.20	0.00
steel transforming, 51000kg	15462	44.5	15.12	0.97	2.10	0.47	0.00
environmental burden of construction activity	P.M.	P.M.	P.M.	P.M.	P.M.	P.M.	P.M.
Total in kg equivalent:	269694	910.2	105.77	92.29	113.92	1.57	21110
multiplierfor normalisation	0.135	7.55	3.60	8.90	27.44	802	1
Eco-costs 2007 (€)	36409	6872	381	821	3126	1261	21110
Note: Total Eco-costs approx. 70.000 (for materials only)							

Table 3.7. Virtual eco-costs '99 for a warehouse, 920 pallets (900 m^2, 10 meters high); calculation according to a standard cost estimate system and the EVR model; the EVR has been based on LCA data, see www.winket.nl

Warehouse, 920 pallets, 900 m2, 10 m high	Value € / m2	EVR	Eco-costs € / m2	Eco-costs € / 900 m2
floor , reinforced concrete, 300 mm thick	170	0.43	73	65.790
steel structure	105	0.28	29	26.460
foundation of steel structure	15	0.43	6	5.805
roof, steel+thermal insulation	40	0.32	13	11.520
Cladding+ insulation (surface.=1,3xfloor area)	95	0.51	s48	43.605
Lighting, heating, sprinklers, etc.	60	0.33	20	17.820
Total	485	0.39	190	171.000

3.6 The EVR and the virtual eco-costs '99 for industrial activities

Similarly to the calculations on the pollution prevention costs of materials (Table 3.1), calculations have been made on these costs of industrial activities. These calculations are based on an extensive measurement programme on the emissions of industrial sectors in The Netherlands. Furthermore the eco-costs of energy have been calculated

on the basis of the energy consumption of these industrial sectors, and the eco-costs of depreciation and labour have been estimated on financial data on these sectors.
The results of the calculations are provided in Table 3.8.
To calculate the data of Table 3.8, the measured industrial pollution in 1995 (VROM, 1997) has been compared with general statistic data on these industries for 1995 (CBS, 1997).
Basically, the EVR data of Table 3.8 form the link between the LCA-based approach and the "input-output table" based approach of macro economic environmentalists.

Table 3.8. The eco-costs of industrial activities, and the corresponding EVR; The Table is based on Added Value and emissions of the sector itself, excluding sourced materials.

INDUSTRY (CBI-code)	pol. prev. costs '07 in 10^6 € of 1995	eco-costs 2007 in 10^6 € of 1995	Added Value in 10^6 € of 1995	EVR
Food industry (15,16)	614	1014	9031	0.11
Textile- en clothing-industr.	76	186	838	0.22
Leather industry (19)	8	18	81	0.22
Wood industry (20)	37	87	415	0.21
Paper industry. (21)	398	613	1386	0.44
Printing industry (22)	150	546	3404	0.16
Oil industry (23)	2144	2405	890	2.70
Basic chemicals ind. (2412-	1851	2471	4677	0.53
Fertiliser industry. (2415)	283	incl. basic chem	incl. basic chem	-
Agriculture chemicals (242)	6	12	44	0.28
Coatings- and ink-industry (243)	8	56	420	0.13
Pharmaceutical industry (244)	33	not available	not available	-
Detergents industry (245)	7	41	307	0.13
Other Chemicals (246)	55	not available	not available	-
Fibre industry (247)	111	not available	not available	-
Rubber industry (251)	12	212	1332	0.16
Converting industry plastics (252)	58	incl. in rubber	incl. in rubber	-
Building materials (26)	617	842	1708	0.49
Basic metals industry (27,231)	1712	1955	1840	1.06
Metal products industry (28)	100	441	2766	0.16
Machine- en equipment ind. (29)	40	391	3065	0.13
Electrical industry (30-32)	109	682	4517	0.15
Automotive industry (34)	69	381	1706	0.22
Shipyards (351)	47	incl. auto ind.	incl. auto ind.	-
Instrument- and optical ind. (331)	9	46	312	0.15

3.7 Discussion

3.7.1 Eco-efficiency

Since the EVR links the 'value' with the 'ecological impact', the EVR is also a parameter for the eco-efficiency as defined by the WBCSD. We propose, however, the following equation, which is more in line with the definition of efficiency in other sciences (see Figure 3.4):

(3.5) eco-efficiency = (value – eco-costs) / (value), or eco-efficiency = 1 – EVR

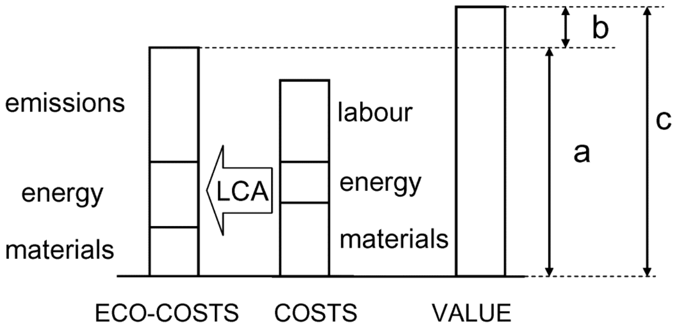

Figure 3.4. The definition of eco-efficiency in science : eco-efficiency = *b/c*.

Note that the eco-efficiency is:

- negative when the eco-costs are higher than the value, or where EVR > 1
- 0% when the eco-costs are equal to the value, or when EVR = 1
- 100% when there are no eco-costs, or when EVR = 0.

3.7.2 Accuracy

Rather than accuracy, practical choices on system characteristics and system boundaries are the major concern in an LCA (What is included? Which processes? Industrial averages or best practices?, etc.). Sensitivity analyses showed that these choices are dominant in the EVR calculation model (as they are in other LCA-based models applying a single indicator).
The 'value' in the equation is not as vague as many non-specialist would suspect: in the EVR model, the value is defined as the 'sales price' within the business chain and the 'fair price' in the consumer market, which are quite well determined in practice.

A way to analyse the topic of accuracy is to study the variance of LCA data and sales prices within Europe. Two examples:

- For a specific design of a solid or corrugated board box, the best practice versus the worst practice in terms of 'clean production' differs more than a factor four within Europe. The value (price) is in this market segment hardly higher than the production costs. Within Europe these production costs differ not more than approximately 20%. Therefore the value is quite accurate in comparison with the eco-costs.
- For electricity, production in Holland is a factor 2 cleaner than in Portugal. In countries with hydroelectric power as Norway, emissions are a factor 100 less than in Holland. (Rombouts, 1999). Prices, however, differ not more than 30% within the EC. Again, the value is quite accurate in comparison with a single indicator for emissions.

Emissions of energy production vary enormously with the choice of type and grade of the fossil fuel. In the eco-costs model, energy from fossil fuels is replaced by renewable energy, rather than preventing the emissions from these fossil fuels. The accuracy of the costs estimates for renewable energy systems is estimated at 30%, which is by far better than the spread in emissions of energy from fossil fuels.

The fact that the EVR is dimensionless means that the EVR model is relatively robust with regard to exchange rate fluctuations or inflation of currencies.

3.8 Call for Comments

1. In the EVR model, economic allocation is applied in two cases:
   a. When subsystems are shared between many product types and the physical relationship is not determined by one parameter, like mass, volume or time. The EVR model then applies formula (3.7)
   b. When there is a linear relation between value and a physical parameter like mass, volume or time. In these cases, there is no difference between the 'physical proportion' and the 'economic proportion'.
   We feel that the use of economic allocation should be defined better than 'only where physical relationship cannot be established'. We suggest a list of criteria which must be fulfilled to allow economic allocation, like:
   - relative stable prices in a transparent, free, and open market and
   - a linear relationship between value (price) and mass, volume or time
   Are there any suggestions to complete this list? Are there any comments?[23]
2. In the EVR model, the 'bonus' of open loop recycling is allocated to the 'new product', and consequently not to the 'old product' (otherwise recycling is counted twice). Our choice was grounded on methodological aspects (avoiding endless loop systems and avoiding the methodological consequences of the build-up of materials in the cycle for products with a long life time).
   We are aware of the fact that our approach is not in line with the current practice of allocating the benefits of recycling to the 'old product'.
   Are there any comments on our choice, and/or are there any suggestions for an alternative (hybrid?) solution for allocation?[24]

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[23] CLM at the Leiden University has reacted with a proposal to make economic not the 'last choice', but the 'preferred choice'. They made a long list of criteria for economic allocation and how to deal with economic allocation specific situations (Guinee, 2004)

[24] Nowadays (2009) it is common practice in LCA to take the bonus of recycling at the beginning of the production chain.

4 Recycling and Cradle to Cradle[25]

4.1 Abstract

This chapter deals a new model for the End of Life (EoL) stage of complex products.

Cradle to Cradle, 'Design for Recycling' and dematerialization by enhancing the durability of products, are major aspects of the quest for sustainable products. This chapter presents an LCA based model for the integrated analyses of the product chain, its recycling systems and its waste treatment systems at the 'End of Life' stage. The model is an extension of the EVR model, but can also be applied to other life cycle interpretation systems, since the model as such is not restricted to the use of the eco-costs as a single indicator.

The model has been developed to evaluate the design alternatives of complex products such as buildings and cars. These products comprise several subsystems, each with its own special solution at the End of Life stage: extending of the product life, object renovation, re-use of components, re-use of materials, useful application of waste materials, immobilization with and without useful applications, incineration with and without energy recovery, land fill.

Since complex product systems always comprise a combination of these design alternatives, a methodology is given to calculate and allocate the eco-costs of the total system in order to select the best solution for sustainability. The methodology is characterized by:

- a main allocation model of the recycling flow based on physical relationships
- a strict separation of the market value, the costs and the eco-costs in the system
- a main allocation model for extension of lifetime based on 'depreciation of eco-costs', parallel to economic depreciation.

[25] The original title was: "Allocation in recycle systems: an integrated model for the analyses of eco-costs and value." Published in Int. J. of LCA (Vogtländer, 2001,C).

4.2 Introduction: current issues with regard to the End of Life stage of products

4.2.1 Complexity

The End of Life (EoL) stage of products is a rather complex stage. Products are collected and dismantled, materials are separated and upgraded, waste is incinerated or dumped, toxic materials are immobilized or incinerated. In terms of the LCA, it is a problem that materials of products are combined with materials of other products, which causes fundamental problems with regard to allocation (Klöpffer, 1996, Ekvall, 1997).

The economics in the End of Life stage is rather complex as well, since products and materials in the End of Life stage often have a negative market value (price) as such. The activities, however, to recycle these products and materials in an environmentally correct manner, have a positive added value for our society as a whole. This results in a situation where the 'free market' has to be restricted in many ways by governmental regulations (e.g. prohibition of dumping certain materials and/or products in land fills), and where the government has to force industry to recycle their products in a correct manner.

In terms of the EVR model (see previous chapter for a short description of the model), the aforementioned complexity means that:

- the allocation model of the End of Life stage has to be defined in an unambiguous way
- the 'value' system in the End of Life stage has to be determined.

4.2.2 Three common ways of looking at the End of Life of products

To unravel the complexity, we may distinguish 3 common ways of looking at the EoL (the 3 EoL paradigms):

- 'The cycle'
- 'The chain'
- 'The cascade'

'The cycle' is depicted in Figure 4.1, being the idealists' way of "how it should be": when 100% of the products and/or materials are recycled, all problems of materials depletion and land fill are resolved. It is Cradle to Cradle.

Modelling the End of Life as one single recycle loop, however, does not cope with two important aspects of the reality:

- 'the second law of thermodynamics', requiring an 'upgrading' activity and requiring 'bleed flows' to cope with degradation, contamination and dilution of materials within the loop.

- 'The many lives of recycled materials', i.e. materials do not stay in one product loop, but switch to other product loops ('cascading' down to other life cycles).

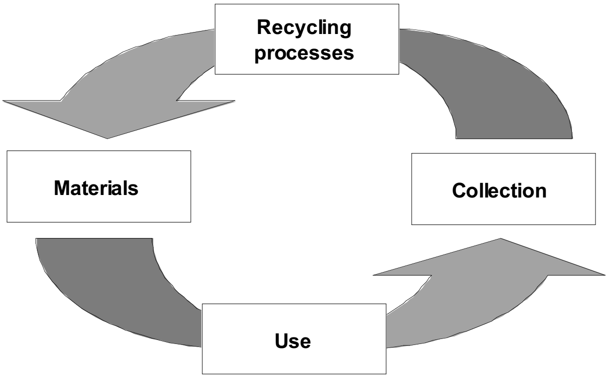


Figure 4.1. The End of Life system from the point of view of idealists: 'the cycle'

'The chain' is depicted in Figure 4.2, being the way product designers and engineers approach the problem of EoL. The recycling systems as such (after the separation step) are normally not included in the product analyses. 'The chain' is the way EoL is configured in design tools such as Simapro, Ecoscan and in the EPS system.

The main focus within this paradigm is twofold:

- try to apply recycled materials for construction elements of the new product or structure
- make it technically feasible (easy) to disassemble or dismantle the product or structure: 'design for recycling' (the 'separation' step in Figure 4.2).

Depicting End of Life as 'a chain' does not cope with two important aspects:

- the recycling activities as such cannot be analysed (alternative systems for transport and upgrading after the separation step), since recycling systems normally combine many 'chains'
- the sense or nonsense of recycling activities as such, with regard to the general subject of sustainability, cannot be analysed (questions such as: is it wiser to recycle a certain type of plastic, or burn it?).

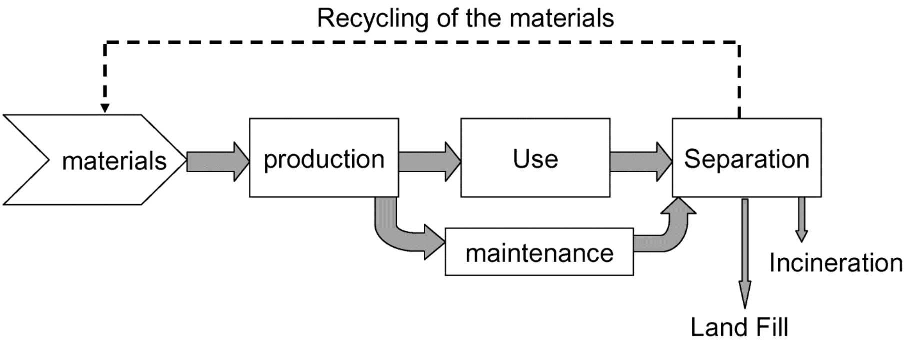


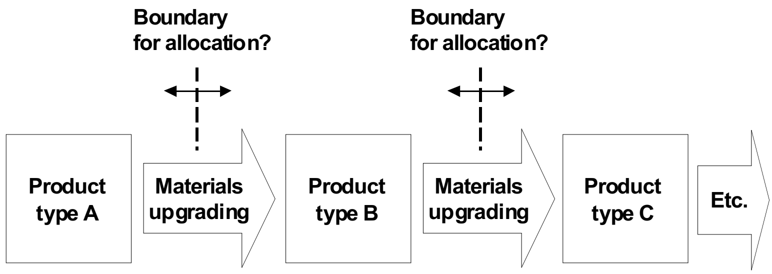
Figure 4.2. The End of Life system from the point of view of product designers: 'the chain'.

'The cascade' is depicted in Figure 4.3, being the way most of the business managers as well as LCA-specialists approach the problem of End of Life. The main focus within this paradigm is "can we do something useful with the old product, or, do we have a 'second life' for the old materials". In the strict sense, 'the cascade' is not a form of recycling, but rather a form of re-use of the degraded materials itself (examples are waste paper, fly-ash, crushed stones and crushed concrete).
The cascade is regarded as the fundamental way to optimize the use of resources (Sirkin, 1994).

The cascade has triggered many proposals and debates among LCA-specialists on the subject of allocation:

- Has the environmental burden related to use of virgin materials to be allocated to the first product only, or has this environmental burden partly to be allocated to the second and third use for products as well? (In the Netherlands this allocation to a second and/or third product life is called 'estafette method', referring to relay races.)
- Given the fact that the End of Life activities (such as separation, transport and upgrading) are causing environmental burden, which of these activities have to be allocated to which product?

Figure 4.3. The End of Life system from the point of view of LCA experts: 'the cascade'.



4.2.3 Order of Preferences of End of Life solutions in The Netherlands ('the Ladder of Lansink')

Given the complexity of the EoL systems, the government of the Netherlands adopted an order of preferences of EoL solutions on which to base the governmental policy. This is a sort list of five EoL solutions, the so-called 'Ladder of Lansink':

- re-use of the product (example re-usable crates for transport of consumables)
- re-use of the materials of a product (example recycling of glass and metals)
- incineration with energy recovery
- incineration without energy recovery
- landfill.

This order of preferences for policy making of the Dutch government was implemented in 1979, and is still the basis for decisions on regulations, legislation, taxation and subsidies.

Designers of a so-called *Product-Service System* (PSS) think along the same lines:

a) try to bring the recycling activity within the PSS and "close the loop" (create 'the cycle')
b) when a) is not a practical solution, create 'the chain' with maximum materials recovery
c) when b). is not feasible (because of severe degradation) try to create 'the cascade' + incineration with energy recovery
d) incineration without energy recovery and landfill has to be avoided.

Although the list of preferences has its basic logic, and although it served successfully as a catalyst for Dutch policy making for two decades, the need for a better system is felt under a vast majority of the people involved:

- there is a need for a more refined list of preferences
- there is a need for a calculation model to check which of the EoL systems on the list is the best practical solution for a specific case in terms of sustainability[26].

In Section 4.3 such a new refined list of preferences is proposed, and it is shown how to make calculations on the eco-costs and the EVR in the chapters thereafter.
First, an overview of the existing theories on allocation in cascade systems will be provided in the next chapter.

4.3 Existing theories for allocation in cascade systems

As mentioned in Section 4.2.3, the main debate on allocation of EoL activities concentrates on 'the cascade' of Figure 4.3. The main question is where to allocate the environmental burden related to the primary production of materials, the recycling activities, and the final waste treatment.

The classical approach of LCA practitioners of separating the assessment of the product chain and the assessment of the recycling system is inappropriate: innovative designs of product-service systems require an integrated assessment of both the product chain and its recycling systems. This becomes even more relevant in cascade systems.

The norm ISO 14041 provides a framework on how allocation problems should be tackled. It describes a three step procedure with regard to allocation[27].

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[26] Note that, for a specific case, the sequence of preferences of the best practical solutions in terms of sustainability can deviate from the general sequence. An example is the best choice of transport packaging for medium distances , where re-use of the product (the re-usable crate system) is less favourable than re-use of material (the board from recycled paper system) because of the extra return transport of the empty crates.

[27] Allocation is defined in this ISO as: partitioning the input and/or output flows of a process to the product system under study. For an extensive analysis on the subject of allocation, see (Frischknecht, 1998).

As a first step, allocation should be avoided where possible (by dividing the process in sub processes or by expanding the product system). As a second step, when allocation cannot be avoided, allocation has to be done in a way which reflects an underlying, causal, physical relationship.
The third step is about 'other relationships' such as economic value.
Some authors argue that economic allocation cannot be avoided here, since neither of the two first steps are feasible, and since the boundary line in Figure 4.3 always leads to arbitrary choices (Lindfors, 1995, Guinée, 2002, Werner, 2000, Ekvall, 2000). The question is, however, whether economic allocation, based on the - heavy fluctuating - market prices of recycled materials, will lead to better results than the simple methods which were originally proposed: "shifting the secondary materials outside the system boundaries" (Klöpffer, 1996), or the "simple cut-off method", where a product made out of primary materials carries the environmental burdens of those primary materials and a product made out of secondary materials carries the environmental burdens of the recycling activities of those secondary materials (Ekvall, 1997).

The EVR model provides a method for economic allocation, but economic allocation can only be applied when specific criteria have been fulfilled (see Chapter 3):

- Relatively stable prices in a transparent, free, and open market.
- A linear relationship between market value (price) and mass, volume and/or time.

It has to be emphasized here, that these criteria are not fulfilled for the economic allocation models which have been proposed by CML (Guinée, 2002), since:

- Prices for products such as scrap and waste paper are highly volatile (unstable).
- The markets for these waste materials are highly influenced by governmental policies.
- There is no simple, linear, relationship between market value (price) and mass.

The economic allocation model which has been proposed by CML (Guinée, 2002) is characterized by the fact that the boundaries for allocation shift with the market price of the waste materials. The EoL activities are being allocated to the next product, when the next product pays for the waste (when the waste material has a positive value). This system is depicted by the example on 'a house to be demolished and processed into road building material', see Figure 4.4.
In Figure 4.4, the environmental burden of the activities in the grey blocks is allocated to the road, the environmental burden of the other activities is allocated to the house.

In Figure 4.4 four situations have been depicted. Quoted from (Guinée, 2002):

- Variant A: waste flow value positive from building, and hence even more positive after processing and in road structure.
- Variant B: waste flow value zero from building, value positive after processing and in road structure.

- Variant C: waste flow value negative from building, value negative after processing but positive in road structure.
- Variant D: waste flow value negative from building, value negative after processing and the road structure has a waste management function as a co-product.'

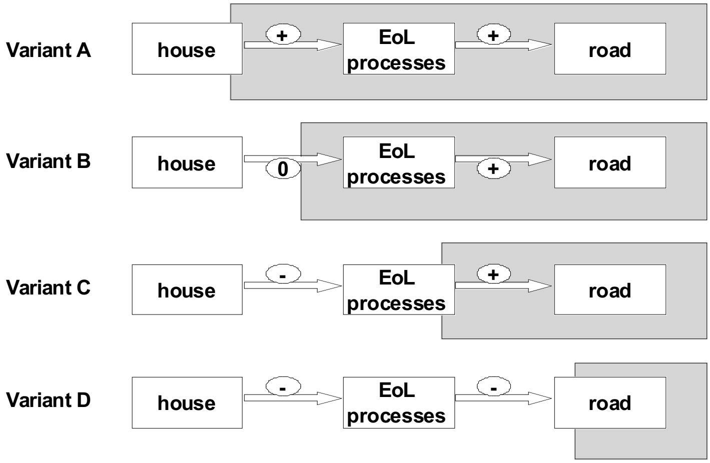

Figure 4.4. The CML allocation model for the situation where materials of a house after demolition are used in road construction , from (Guinée, 2002), simplified.

The basic idea of such an allocation model is that 'the house' has to benefit from the fact that there is a useful appliance of its waste (the designer of the house should be influenced by the LCA to use materials which can be re-used in other products or structures).
The 'estafette' allocation model (estafette = relay race) of (Seijdel, 1994) and the ISO option of 'the number of subsequent uses' (ISO Section 6.4.3) have the same intention: taking the full burden away from the first product.

The main disadvantage of the allocation model of CML is that the value of the waste material is not known at the moment that the house is designed and built: often more than 40 years before the moment of demolition! So at the design and building stage of the house, the boundaries for allocation are not known, which is a rather unpractical situation.
There is also a methodological flaw in the CML system, caused by the fact that it is often the 'bundle of costs and benefits', and/or the governmental regulations, that influences the economic decisions. It is not the price of the waste materials as such which influences the economic decision.
In the example of the house, the reason for demolishing a house is often the fact that the value of the ground area is more than the value of the house. The EoL activities are then a co-product of another activity: project development. The EoL activities are 'subsidized' then by the main product (being the creating of ground area). The value of the waste is hardly influencing the economic decision in such cases. So there is no reason at all to give the economic value of waste materials an important factor in the allocation model.

4.4 The End of Life system of the EVR model and a new order of preferences of EoL solutions (the Delft Order of Preferences)

In Section 4.2 it was concluded that EoL systems are complex, and the three paradigms (the cycle, the chain and the cascade) each cover only a part of the real practice.
In Section 4.3 it was concluded that the existing proposals for EoL allocation, based on the market value of recycled materials, do not fit the reality.
Therefore, a methodology has been developed, which:

- Reflects the underlying, causal, physical relationship (step 2 of ISO 14041) of the materials flow in the recycling markets.
- Can be regarded as an enhancement of early proposals in this field (Klöpffer, 1996, Ekvall, 1997, Kim, 1997).
- Keeps the environmental burden, the market value and the costs in the chain strictly separated.
- Deals not only with recycling, but also with enhancement of the lifetime of a product.

The way that the EVR model deals with End of Life and Recycling is depicted in Figure 4.5.

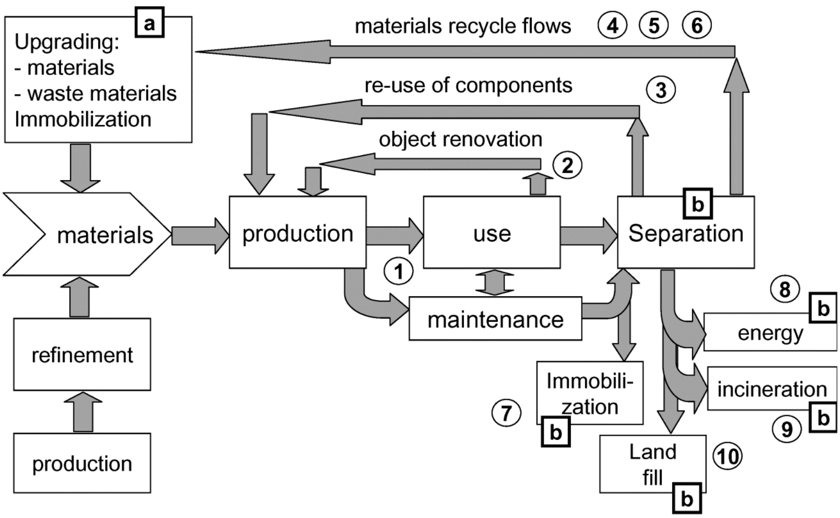

Figure 4.5. The flow of materials in the Life Cycle.

This figure depicts the major types of End of Life treatment and types of recycling. It is developed to describe and analyse the various kinds of complex modern life cycles of products, buildings, manufacturing plants, civil structures, etc.
The numbers in Figure 4.5 relate to the "Delft Order of Preferences", a list of the 10 major systems for End of Life used for structured and systematized analyses of

(combinations of) design options[28]:

1. Extending of the product life
2. Object renovation
3. Re-use of components
4. Re-use of materials
5. Useful application of waste materials (compost, granulated stone and concrete, slag, etc.)
6. Incineration with energy recovery
7. Immobilization with useful application
8. Incineration without energy recovery
9. Immobilization without useful application
10. Land Fill.

It is important to realize that for big, modular objects (such as buildings), there is not 'one system for End of Life' but in reality there is always a combination of systems.
The two basic rules for allocation in the EVR model are:

- Costs and eco-costs of all activities marked with 'b' are allocated to the End of Life stage of a product (transportation included).
- Costs and eco-costs of all activities in the block marked with 'a' are allocated to the material use of the new product (so are allocated to the beginning of the product chain).

There are many reasons to allocate the activities in the block marked with 'a' to the new product, and the activities in the blocks market 'b' to the old product. Three major arguments:

- Physical tracing of recycled material flows between the "separation step" and the "upgrading step" is often impossible (e.g. for recycled materials such as metal scrap and waste paper there is a global trade with large stocks of several grades, so there is no direct physical relationship for those materials between the old product and the new product).
- The processes to upgrade or blend the different grades of recycled materials are often directly related to the use in the new product, sometimes these processes are even integrated in the making of the new product (e.g. paper from recycled paper mills, steel from the Basic Oxygen Steelmaking process, etc.).
- For products with a long lifetime, other allocation models lead to wrong conclusions (Gielen, 1999)[29].

[28] In 2007 the order of numbers 6-9 was slightly changed (incineration was placed higher than immobilization, because of the positive effects in LCA calculations)

[29] Example: when the average lifetime of a car is 10 years, and when the production of cars has doubled in the past ten years, 45% recycled steel can be used in new cars when 90% of the steel of old cars is recycled. It is obvious that the fact that we need 65% virgin material for new cars is more relevant for our society in terms of material depletion and CO_2 emissions, than that we recycle 90% of the old cars.

In fact, the allocation of the activities of block 'a' to the new product is well in line with the allocation procedure in ISO 14041. The recycling activity has been split in subsystems type 'a' and type 'b' (step 1 in the procedure) and the subsystems type 'b ' have been allocated to the old product and the subsystems type 'a ' have been allocated to the new product. This is according to the physical relationship, in line with the ability to trace materials flow in the recycling loop.

In line with the aforementioned allocation strategy, the 'bonus' to use recycled materials is taken at the beginning of the product chain, where the new product is created. Material depletion is caused here when 'virgin' materials are applied, material depletion is suppressed when recycled materials are applied.

The eco-costs of materials depletion are defined by the costs of the fraction 'virgin' materials, $(1 - \alpha)$, which are used for the new product. In formula:

$$\text{(4.1)} \qquad \text{eco-costs of materials depletion} = (\text{eco-costs of 'virgin' materials}) \times (1 - \alpha)$$

Where α is the fraction of used materials for the new product which stems from recycled material (when upgrading is required, after the upgrading step). See Section 3.4.3.

The 'separation' block in Figure 4.5 comprises a chain of activities for most products. To end up with the best grade and the best purity of recycled materials, the separation step of products has normally to be organized in at least three steps:

- Dismantling of the product in components.
- Demolishing the components (in a shredder).
- Separation of the output of the demolishing step (by a magnet, by eddy current, by air flow, etc.).

For buildings the same principle applies: dismantle the building first (taking out the wood, glass, cables, metals, etc.) prior to demolishing the building. The quality (purity) of recycled materials is then much better, compared with the 'classical method' (demolishing as the first step and separation afterwards).

This has two consequences for the future building industry:

- The design of a building has to be such that the building can easily be dismantled in order to be able to separate the several building materials; this 'design for recycling' is common practice now for consumer electronics and cars.
- Low capacity, transportable, processing equipment for separation and crushing at the site of the old building is to be preferred instead of processing in big, centralized, separation plants, in order to avoid contamination and degradation of recycled materials during the materials handling, transport and storage prior to processing.

4.5 The eco-costs of End of Life and recycling activities

4.5.1 The eco-costs of End of Life of a product

All of the activities of Figure 4.5 have their emissions, use of energy and use of additional materials (e.g. the equipment which is used), so all of these activities have eco-costs. As it has been mentioned in the previous section, eco-costs of all activities (transportation included) marked with 'b' are allocated to the End of Life stage of the old product. In formula:

(4.2) eco-costs of EoL = Σ (eco-costs of activity type 'b')

Note that there is no 'estafette (relay race) effect' in the allocation model of the EVR model because of the clear division between activities 'b', to be allocated to the EoL of the old product, and activities 'a', to be allocated to the new product.

With regard to the summation of eco-costs according to equation (4.2), the analysis of two blocks of activities in Figure 4.5 needs extra attention:

- Incineration with energy recovery (block number 8).
- Land fill (block number 10).

For incineration with energy recovery, there is a surplus of energy in the Life Cycle, which results in negative eco-costs of energy in equation (4.2), since energy is 'exported' to other products.

For land fill it has been decided by the Dutch government that land fill is not a sustainable solution for waste treatment, and therefore has to be avoided (prevented)[30]. Consequently, the EVR model introduces the 'eco-costs of Land fill', being the costs of prevention of land fill. The 'last' main prevention measures for Land fill to reach the target (the 'marginal prevention costs') are:

- Making compost of bio waste: processing costs 90 € per 1000 kg.
- Incineration of domestic waste in an environmentally acceptable way: processing costs 118 € per 1000kg.
- Recycling building materials: extra costs of separating materials at the EoL (partly 'dismantling' instead of total 'demolition' of the structure) is less then 118 € per 1000kg in most cases.

Consequently, the 'eco-costs of land fill' have to be set at 118 € per 1000 kg, being the costs of these marginal prevention measures to reach the target.
It has to be mentioned here that the target setting for land fill in the Netherlands has been a political choice: the Dutch society is apparently willing to pay about 118 € per

[30] The governmental policy in The Netherlands is to restrict land fill. In 1996 14% of the total waste flow was land fill, the target for 2010 is 4% ! Land fill for toxic materials is forbidden by law.

1000 kg in order to minimize land fill to the level of about 4% of the solid waste. In fact, the 4% target is a result of what is considered as feasible in the technical/ economical sense[31]. It is obvious that regions which are less densely populated will tend to take less expensive measures to prevent land fill (e.g. they will not be prepared to invest in incinerators and large scale compost production). Or should these regions apply the 'best practices'?

4.5.2 The eco-costs of using recycled materials for a product

Scrap metal, waste paper, waste glass, waste plastics, waste wood, etc. are regarded as the source for 'recycled' materials (as metal ore, pulp for paper, etc. is the source for 'virgin' materials). The eco-costs of the processes to make the new material from the waste material is allocated to the material which is used in the new product. In the EVR model, the eco-costs of materials of a new product are calculated according to:

(4.3) $$\text{eco-costs of materials} = \Sigma\,(\text{eco-costs of energy + pollution prevention costs})_{\text{upgrading of recycled materials}} + \Sigma\,(\text{eco-costs of materials depletion + eco-costs of energy + pollution prevention costs})_{\text{virgin materials}}$$

In most cases virgin materials require more energy and cause more pollution than recycled materials (e.g. metals and glass). See also Table 2.1[32].

For situations of combined material production, such as in the Basic Oxygen Steelmaking process, equation (4.3) can be combined with equation (4.1) and written in the form of equation (4.4):

(4.4) $$\text{eco-costs of materials} = \Sigma\,(\text{eco-costs of 'virgin' materials}) \times (1 - \alpha) + \Sigma\,(\text{eco-costs of energy + pollution prevention costs})_{\text{processing of all materials}}$$

Where α is the fraction of material for the new product which stems from 'recycled' material (after the processing step!).

4.5.3 The eco-costs of recycling

In the previous Sections 4.2 and 4.3, the way of calculation and allocating eco-costs of the recycling loop has been dealt with. This approach was focussed on the 'eco-costs of a product'.

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[31] This situation is different from the setting of norms for emissions in Chapter 2. For emissions, the Dutch government has based their norms on the 'negligible risk level' for concentrations (in air and in water) and the corresponding 'fate analyses' (the link between concentration and emissions). Although there are many scientific disputes over these kind of calculations, they are less arbitrary than just the 'political will'.

[32] Note that in most of the LCA data on materials, the pollution data include pollution from the use of energy. In those cases energy must not be counted extra in the formula for the total eco-costs, to avoid counting energy twice. See Chapter 3.

In this section we will deal with the subject of the 'eco-costs of recycling', and the ability to analyse (closed loop) recycling systems as such.

To calculate the eco-costs of recycling the following activities are included, see Figure 4.5.

- All activities type 'b', including the required transport and storage.
- All activities type 'a', including the required transport and storage.

For calculation (comparison) of recycling systems, the following assumptions are made:

- The recycling system is 'closed loop' (the materials of the EoL of a product are recycled and used for a new product of the same type).
- Time (material hold-up) is not taken into account.
- When a material fraction α in the new product stems from recycling, a material fraction ($1 - \alpha$) in the new product stems from 'virgin' material, and a fraction ($1 - \alpha$) ends up in one of the following EoL systems (see the "Delft Order of Preferences", Figure 4.5):
  - Incineration with energy recovery
  - Immobilization without useful applications
  - Incineration without energy recovery
  - Land fill.

Note that degradation of the product is taken into account by the 'bleeding' of a small fraction to either of the following EoL systems: Immobilization without useful appliances, incineration, or land fill. This 'bleed' of material will lead to virgin material entering the life cycle loop and will keep the grade in the recycling loop at an acceptable level.

The eco-costs of a recycling system are 'virtual', since the aforementioned assumptions hardly exist in real life (recycling systems are not closed loop in the real sense of the word).
The total eco-costs of recycling are defined as (see Figure 4.5 for activity type 'a ' and type 'b'):

(4.5) eco-costs of recycling = Σ (eco-costs of activity type 'b') + Σ (eco-costs of activity type 'a')

When a classical analysis is made of recycling systems as such (without integration with product chains), the benefits in terms of *avoided eco-costs* might be taken into account. These benefits relate to the fact that less 'virgin' materials are used (resulting in less materials depletion and normally less pollution and less use of energy at the materials production stage).

When α is the fraction of material of the new product which stems from 'recycled' material, the 'net eco-benefit of recycling' can be defined for the total recycling system as:

$$\text{'net eco-benefit of recycling'} = \Sigma \{(a + b + c) - (d + e) + f\} \times \alpha \tag{4.6}$$

Where:
a = (eco-costs of materials depletion)$_{\text{at 100\% virgin material}}$
b = (eco-costs of energy)$_{\text{at 100 \% virgin material}}$
c = (pol. prev. costs)$_{\text{at 100\% virgin material}}$
d = (eco-costs of energy)$_{\text{at 100\% recycled material}}$
e = (pol. prev. costs)$_{\text{at 100\% recycled material}}$
f = (eco-costs of immobilization, incineration or Land Fill).

The 'net eco-benefit of recycling' ranges from zero ($\alpha = 0$, no recycling) to a maximum ($\alpha = 1$, 100% recycling)[33], where $\alpha = 1$ is not feasible because of the Second law of thermodynamics.

The purpose of equation (4.6) is to bring the positive effect of re-using materials within the boundary limits of the analysed recycling system. When total loops or total systems are to be analysed (when the product chains are included within the boundary limits), equation (4.6) must not be used, to avoid double counting of the 'avoided eco-costs'.

4.6 The Value and the EVR of EoL and recycling systems

4.6.1 The value in the recycling loop

The economics in the End of Life stage and the economics of recycling are rather complex, since, in most cases, products and materials in the End of Life stage have a negative market value (companies who take away discarded products are paid for it). People can earn money by keeping these products in stock, resulting in an enormous hold-up of discarded products world wide, and resulting in a certain pressure to get rid of it in an illegal way. Therefore, the free market of discarded products has to be restricted in many ways by governmental regulations (e.g. prohibition of dumping certain materials and/or products in land fills), and the government has to force industry to recycle their products in a correct manner.

The services (activities) to recycle these products and materials in an environmentally correct manner have a positive added value. Within the framework of regulations and

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[33] A similar model is proposed for 'environmentally weighted recycling quotes', to replace the 'material recycling efficiency' used by several member states in the EU, which describes the performance of recycling systems (Huisman, 2000a, Huisman, 2000b).

joint agreements between government and industry (the Dutch 'convenants'), a free market of recycling activities can thrive.
A special problem, however, for the free market of recycling is the fact that recycled materials fluctuate heavily in price. This instability of prices results from the fact that:

- Price speculation in recycled products is cheap (because of the negative investment in stock).
- Some governments in the western economies subsidize processing of waste materials, while others do not (sudden introduction of subsidies, regulations and the like, disturb markets and market prices).
- Some countries in the Far East buy waste materials (such as waste paper) in enormous quantities at one time (transport of waste materials from Europe to the Far East is extremely cheap because of the surplus of empty containers returning to the Far East).

This situation has the following consequences:

- The negative market value of discarded products and materials is indirectly determined by the governmental regulations and levies on waste treatment, which are a result of the 'willingness to pay' to avoid land fill.
- The market value of recycled materials (after upgrading) might be less than the total costs of recycling.
- The recycling activity is economically feasible when the added value of the recycling activity is larger than the added costs.

The analyses of eco-costs, costs and value of recycling chains must be done with great care. The best approach is to keep these 3 elements strictly separated along the chain, avoiding the total picture getting blurred by mixing up the economic and environmental aspects.

The value and the costs along the recycling chain are depicted in Figure 4.6.
The negative market value of a product when it is discarded ('a' at Figure 4.6), is determined by the levy required for land fill. In the Netherlands this levy is set at 120 – 180 € by the Dutch government for non-toxic materials. It is the right governmental policy that this levy has been set slightly higher than the prevention costs – the eco-costs – of Land Fill (118 € per 1000 kg for non-toxic materials, see Section 4.2)[34]: in most cases prevention is more attractive than Land Fill from the economic point of view.

[34] For toxic products land fill is forbidden in The Netherlands: proper treatment of such waste is obligatory.

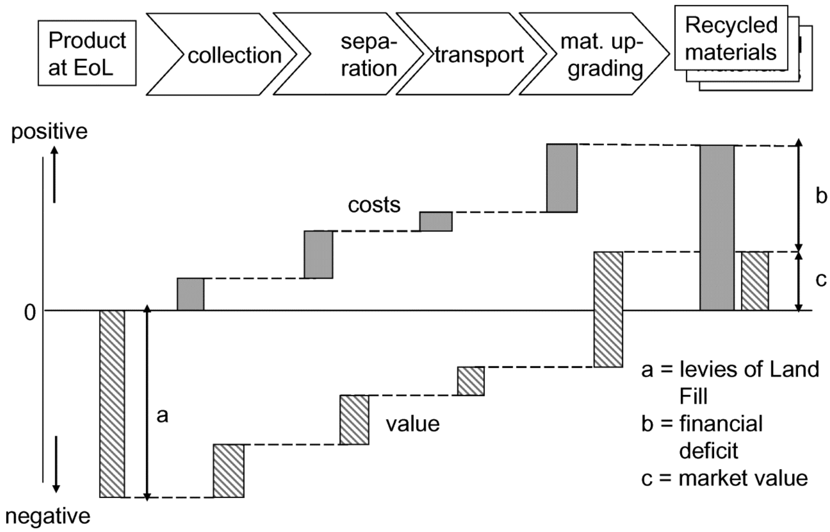

Figure 4.6. The value and the costs along the recycling chain for non-toxic consumer products.

Each step (activity) in the recycling chain adds value as well as costs.
At the end of the recycling chain, we expect a positive market value of the recycled materials ('c' at Figure 4.6). The costs of recycling, however, are often higher than the market value of the recycled materials (this applies to most of the consumer products; waste paper is one of the exceptions). The deficit ('b' in Figure 4.6) at the end of the recycling chain has to be less than the levy 'a' at the beginning of recycling chain (otherwise there is no economic feasibility for recycling).
This deficit 'b' has to be paid in one way or the other, to make the business of recycling profitable. There are 4 major forms of additional payments to the recycling chain:

- The deficit of the recycling is compensated for by a 'waste treatment levy' ('verwijderingsbijdrage') which is paid by the consumer at the moment of purchase of the product; a list of levies for the Netherlands is given in Table 4.1.
- The deficit of the recycling is paid from other sources in the 'bundle of costs and benefits'. (An example is given in Section 4.3: the reason for demolishing a house is often the fact that the value of the ground area is more than the value of the house; the EoL activities are then a co-product of another activity: project development. See also the example in Section 4.7).
- The deficit is paid by the industry involved in the production and trade of the product type (e.g. glass bottles in the Netherlands).
- Subsidies from governments.

	€		€
Televisions	8,00	Grill, microwave	5,00
Portable radios, cassette players, etc	0,00	VCRs	3,00
Light bulbs (energy saving types)	0,14	Domestic appliances	1,00
Ovens	5,00	Freezers and refrigerators	17,00
PCs	9,00	Domestic appliances	1,00
Washing machines	5,00	Car tyres	2,04

Table 4.1. The 'waste treatment levy' ('verwijde-ringsbijdrage') in the Netherlands. (price level 2009).

4.6.2 The EVR model for more advanced sustainable EoL solutions

In the previous sections the EVR model has been described for waste treatment and recycling systems, systems 4 through 10 of the 'Delft Order of Preferences', see Figure 4.5.

In this section we will deal with:

- Extension of the product life (choice of materials, construction type, maintenance systems, etc.).
- Object renovation (e.g. refurbishing of buildings, renovation of buildings, using former building structures and/or foundations).
- Re-use of components (e.g. repair of cars with components of discarded cars).

The essence of these EoL systems is that the life of a product and/or the life of a product component is extended in time. In practice all kinds of combinations of these 3 systems occur in complex products such as office buildings, manufacturing plants, trucks, cars, etc. Therefore the cascade approach (see Sections 4.2.2 and 4.4) is not suitable to tackle the problem of analyzing these types of systems.

The way the extended life solutions are dealt with in the EVR model, is similar to the approach which is used for eco-costs of depreciation of production facilities as described in Chapter 3, and is explained hereafter.

When, for example, the lifetime of a product is extended by 10%, the eco-costs per year are decreased by 10%. If this enhancement of the lifetime is achieved at the same costs and the same value per year, the EVR is decreased by 10%

The underlying assumption is that eco-costs are distributed in a linear way over the lifetime of a product, similar to the depreciation of the costs of a product.

This underlying assumption is also used in Table 4.2, that provides an overview of the eco-costs of an office building. In this table the lifetime of the elements of an office building are used to calculate the eco-costs per m^2 per annum for each element, in order to determine the eco-costs per m^2 per annum of the total. Note that the subsystems in Table 4.2 (building structure, building systems, interior, computer systems) each have different lifetimes.

It is advised to approach the analyses of extended life solutions (with several sub-systems) by calculating the 'eco-costs per annum', applying the following equations:

(4.7) $\quad (\text{eco-costs per annum})_{\text{subsystem}} = (\text{eco-costs / lifetime})_{\text{subsystem}}$

and

(4.8) $\quad (\text{depreciation per annum})_{\text{subsystem}} = (\text{value / lifetime})_{\text{subsystem}}$

Combining (4.7) and (4.8) results in:

(4.9) $\quad (\text{eco-costs per annum})_{\text{subsystem}} = (\text{depreciation per annum})_{\text{subsystem}} \times \text{EVR}_{\text{subsystem}}$

For the total extended life system:

(4.10) $\quad (\text{eco-costs per annum}) = \Sigma \{ (\text{depreciation per annum})_{\text{subsystem}} \times \text{EVR}_{\text{subsystem}} \}$

The meaning of equation (4.10) is that the eco-costs per annum can be derived from the normal costs of depreciation by multiplying it with the EVR of that subsystem. An example of such a type of calculation is given in Table 4.2, for the same office building as given in Table 3.4.

One may argue that equation (4.10) may also be used in situations where the depreciation is not linear, but follows the real market value of a subsystem.

Summary description of an office building (typical)	Investment (€/m²)	Life time (years)	Depreciation (€/m2/year)	EVR	Eco-costs (€/m²/year)
Subtotal construction building structure	630	40	15.75	0.35	5,5
Building systems (elevators, heating, electrical, etc.)	170	20	8.5	0.35	3
Interior (painting, decorating, furniture,etc.)	340	15	22.7	0.3	7
Computer system (one screen per employee at 33 m²)	30	3	10	0.3	3
Maintenance of building and building systems per year		-	15	0.2	3
End of Life:					
Demolition + transport of materials at End of Life	40	40	1	0.5	0,5
Disposal of construction waste (eco-costs 0.1 €/kg)	40	40	1	1	1
Total					23

Table 4.2. Summary of the eco-costs of an office building excluding energy during the use phase; This office building is the same as the office building of the example of Table 3.4.

4.7 Example: a warehouse building

As an example of the EoL and recycling in the EVR model, the concrete floor slab and the steel superstructure (including roof, cladding and warehouse racks) have been analysed for EoL and recycling solutions. The summary of this analysis is given in Table 4.3.

Note: all values in €	0% recycled	95% re-use	Eco-benefit
EoL of Old Building			
Eco-costs of Land Fill	86500	4.325	82.169
Eco-costs of transport	700	716	0
Total eco-costs EoL	87.200	5.040	
		Total eco-benefit EoL	82.169
Materials for new building			
Eco-costs of steel	39.930	1.996	37.933
Eco-costs of concrete floor	30.360	1.518	28.842
Total eco-costs materials	70.290	3.514	
		Total net eco-benefit materials	66.775
		Total net eco-benefit	148.945

Table 4.3. The eco-benefit (in €) of re-use of the steel of the steel structure and re-use of the floor slab of a warehouse; For details see Tables 3.6, 3.7 8.7 and 8.8.

The warehouse is the same warehouse as given in Tables 3.6 and 3.7, and can be summarized with the following characteristics:

- Function: 920 pallet places (900 m^2 , outside height 10 m).
- Concrete floor slab: 650,000 kg, eco-costs 30,089 €.
- Steel, total: 73,000 kg, eco-costs 35,648 €.
- Total eco-costs of the materials of the warehouse, excluding EoL: 70,290 €.

The maximum eco-costs of EoL, e_0 , have been calculated under the assumption that all materials are dumped in a land fill.
The minimum eco-costs of EoL, e_{95} ,have been calculated for 95% re-use, so only 5% of the total weight ends up in land fill.
The net eco-benefit of re-use, allocated to the 'old building' equals: e_0 - e_{95}.

The *avoided eco-costs of materials* in this case entirely depends on:
- The function of the new building which will replace the old building.
- The decision of the architect on the possibilities to re-use parts of the old floor slab and the structure.

Hence the avoided eco-costs of materials, being the net eco-benefit of materials, are allocated to the new building.
Suppose the new building has exactly the same function (a warehouse) and the architect applies recycled steel for 95% of the steel elements, and uses the old floor slab.
For this case the following data has been calculated, and provided in Table 4.3:
- The eco-costs without recycling and re-use.
- The eco-costs of 95% recycling and re-use.
- The net eco-benefit of materials.

Note that, when the design load on the floor slab of the new building is less, the thickness of the floor slab can be less, with consequently a lower amount of concrete required and therefore less net benefit of eco-costs.

In the case that the floor slab has to be demolished and removed, the value and the costs along the recycling chain can be analysed, see Figure 4.6.
Suppose:
- The negative value (the levy) of Land Fill is 110 €.
- The costs of crushing and grinding of the floor slab (including extra transport) is 90 €.
- The value of the granulated material is 10 €.

In this case recycling is economically feasible, since the recycling operation results in an added value of 120 € at added costs of 90 €. In Figure 4.6 the size of 'b' is less than 'a'.
The fact that the value of granulate is less than the costs of the granulate is not ruling the economic decision: this deficit is paid from other sources in the 'bundle of costs and benefits' of the total project.

However, when the size of ‘b’ is more than the size of ‘a’, the recycling operation as such has less added value than added costs, and will therefore not happen in a free market economy. When society insists on recycling in such cases, governmental regulations, levy systems or subsidies are required to make recycling happen.

4.8 Discussion

Although the methodology for recycling and EoL has been developed within the framework of the EVR model, the same methodology can be used in other systems, such as the eco-indicator 95 and 99. This is because of the fact that eco-costs, costs and value are strictly separated from each other. Computer models which are used in the design stage of products, such as Simapro and Ecoscan, are structured according to similar principles, which is in line with the way of reasoning of Ecoinvent (providing LCA data on virgin metals and ‘secondary metals’).

Confusion on the analyses of recycling systems stem from the fact that different parameters in the system are often mixed up:

a) Eco-costs of recycling activities (Equation (4.5)).
b) Eco-benefit of recycling (Equation (4.6)).
c) Costs of recycling activities.
d) Value of recycling activities.
e) Value of the recycled materials.

Note that the EVR calculations as described in Section 4.6 are only allowed for the parameter a), c) and d) of this list, and not for parameter b) and e). Note also that ‘value’ is here the market value.

With regard to the enhancement of the durability of products (Equation (4.10)) it is important to realize that the lifetime of all subsystems of a product has preferably to be the same, when it is not possible to split the product in subsystems at the End of Life. Subsystems with a longer lifetime than the other subsystems suffer from ‘waste of quality’.

However, one has to realize that the End of Life of a product is not a matter of the technical lifetime only, it is also related to value aspects of the product. A product can become obsolete for the user for many reasons (van Nes, 1998):

1. Technical: the product is worn out and no longer functioning properly.
2. Economic: new products have a lower level of ‘Costs of Ownership’ (maintenance, energy, etc.).
3. Ecological: new products have less harmful impact in the use phase (maintenance, energy, etc.).
4. Esthetical: new products have a nicer look, a more fashionable design, a better image (‘feel good factor’).
5. Functional: new products fulfil more functions or fulfil functions better.

6. Psychological: old products have a negative emotional factor (unpleasant history), new products have a positive emotional factor (gift, pleasant history), 'feel good factor'.

To cope with the obsolescence of point 2 through 5, the product design has to be modular. An obsolete module can easily be replaced then by a new one, instead of replacing the whole product. This principle applies also to the design of buildings.

The fact that a product (or its subsystem) can become obsolete before the product is worn out and no longer functioning properly, is the reason that one should take the 'economic lifetime' as lifetime in the LCA calculations (Equation (4.7)), instead of the 'technical lifetime'.
This is the reason why the depreciation of the eco-costs in the EVR model is done in parallel with the economic depreciation (Equations (4.7), (4.8) and (4.9)).

5 Ecoefficient Value Creation[35]

5.1 Abstract

Product designs for the future will need a high value/costs ratio and need to incorporate high levels of eco-efficiency.
Therefore, a new model has been developed to assess the sustainability of products, the EVR model, this model comprises two concepts:

- *Virtual eco-costs* which is a LCA-based single indicator for environmental impact,
- EVR (eco-costs/value ratio), an indicator which shows the de-linking of economy (value) and ecology (eco-costs) of a product or a service.

In this chapter, the advantage of combining analyses of eco-costs and value is considered, and it will be shown how the EVR model can support decision making processes.
The following subjects are analysed:

- The optimisation of a product in the design stage through use of the EVR model (the 'EV Wheel').
- Optimisation strategies for the production and distribution chain of a product (Case: a classical CRT TV).
- The strategic dilemmas relating to marketing of products with low environmental impact (Case: a 'low energy CRT TV').
- An investment policy which lowers the environmental impact of systems, analysed by use of the 'eco-payout time'.
- The EVR model applied to consumer spending: the lifestyle of consumers, and the so-called 'rebound effect'.

At the end of this chapter the consequences are summarized for product development and marketing strategies.

5.2 Introduction: value, costs, and eco-costs

In the transition towards a sustainable society there is a need for products with a low environmental burden (lower emissions and lower use of energy and materials). But we

[35] The original title was: "The EVR model as a tool to optimise a product design and to resolve strategic dilemmas." Published in J. of Sustainable Product Design (Vogtländer, 2001, D).

need a strong economy as well, to support this transition. Such a need for "a new era of economic growth" is described in the preface of "Our Common Future"[36] (Brundtland, 1987). For industry it means that products must create 'value' for customers, whilst producing a low environmental burden.This has been defined as ecoefficiency by the WBCSD (see Annex 5.1, Figure 5.10). The value aspects of sustainability, and the consequences for design and marketing, have been forgotten so far by many environmentalists. Hence the quest for a new approach.

At Delft University of Technology, a model has been developed to assess the so-called *eco-efficiency* of products and services. In this model the ecological burden is monetarized and expressed in one single indicator, the socalled 'virtual eco-costs'[37]. The product life cycles are analysed for three main aspects: the eco-costs, the (economic) costs, and value. The key indicator for sustainability in the model is the *eco-costs/value ratio* – the EVR (Vogtländer, 2001,B). The basis of the EVR is depicted in Figure 5.1. The background of the model is summarized in Annex 5.1.

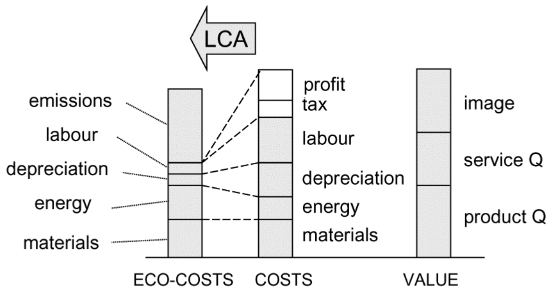

Figure 5.1. The costs, the eco-costs and the value of a product: the basis of the EVR model (*Q* = Quality)

In its broadest sense, the EVR is an indicator that describes the level of de-linking of the economic value and the ecocosts of a product. A high level of prosperity can only be combined with a low level of pollution and material depletion, when products (and services as well) are used with a low EVR, since such products combine a high value with a low level of burden for our earth.

In terms of EVR, the 3 stakeholders of a sustainable society each have their specific roles:

- Industry must develop products and services with a low EVR.

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[36] "The downward spiral of poverty and environmental degradation is waste of opportunities and of resources. In particular it is a waste of human resources. These links between poverty, inequality, and environmental degradation formed a major theme in our analysis and recommendations. What is needed now is a new era of economic growth - growth that is forceful and at the same time socially and environmentally sustainable." (Preface of "Our Common Future" (Brundtland, 1987; page xii).

[37] The concept of the 'virtual eco-costs' is slightly different from the concept of the 'external costs'. External costs are related to damage to our environment. The virtual eco-costs are related to the (marginal) prevention costs, which are required to bring our economy into a state which is sustainable. What both type of costs have in common, is that they are not incorporated in the current economic costs of products and services.

- Consumers and citizens should have a life style with preferences for products and services with a low EVR (e.g. they should spend their money on other activities than travelling by air).
- Governments should encourage products, processes and lifestyles with a low EVR (e.g. labour intensive activities) and should restrict products, processes and life styles that have a high EVR (e.g. energy intensive activities, activities with high toxic emissions).

But what does it mean for the decisions the stakeholders have to make? How do they know they have made the right choice in terms of sustainability?

5.3 The EV Wheel for product design

In the United Nations Publication Paper "Ecodesign – A promising approach to sustainable production and consumption", (Brezet & van Hemel, 1997), the LiDS wheel (LiDS stands for Life cycle Design Strategy) is proposed as a tool to define the design strategy for sustainable products. This LiDS wheel is a qualitative tool, and at Delft University of Technology many attempts have been made to make it quantitative. The EVR model provides the opportunity to do so. The concept of the wheel has been adapted to incorporate the main aspects of the value (product quality, service quality, and 'image & design') of a Product Service System (PSS). A new name has been chosen: the Eco-costs & Value Wheel, in short the EV Wheel.

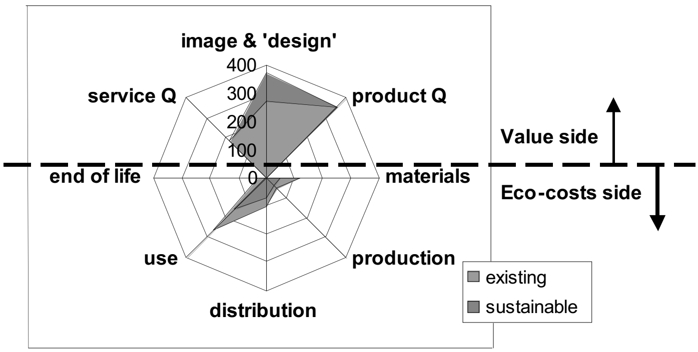


Figure 5.2. The Eco-costs & Value Wheel (EV Wheel). The value and ecocosts are given for a 28" television (tentative) in €: 400 € at the outer ring and 0 € at the centre (Q =Quality).

The EV Wheel is meant to provide a quantitative overview of the eco-efficiency of a product, a service or a combination of both (a PSS).

The value side of the EV Wheel is based on the three main elements of value (product quality, service quality, and 'image & design'). These main elements can be derived from the '8 dimensions of quality' of Garvin (Garvin, 1988):

1. Performance or the primary operating characteristics of a product or service. Example: for a car: its speed and acceleration; for a restaurant: its good food.
2. Features or the secondary characteristics of a product or service.
   Example: for a restaurant: it's linen tablecloths and napkins.
3. Conformance or the match with specifications or pre-established standards. Example: for a component: it's whether this component is the right size; for a restaurant: it is whether the meat is cooked according to your request (e.g. medium or rare).
4. Durability or product life.
   Example: for a light bulb: it's how long it works before the filament burns out.
5. Reliability or the frequency with which a product or service fails.
   Example: for a car: it's how often it needs repair; for an airline: it's how often flights depart on schedule.
6. Serviceability or the speed, courtesy and competence of repair.
   Example: for a car: it's how quickly and easily it can be repaired; for a mail order house: it's the speed and the courtesy with which an overcharge is corrected.
7. Appearance/aesthetics or fits and finishes.
   Example: for a product or service: it's its look, feel, sound, taste or smell.
8. Image or reputation.
   Example: for a product or service: it's the positive or negative feelings people attach to any new products, based on their past experiences with the company.

For the purpose of the EV Wheel, we group the 8 dimensions as follows in 3 aspects of value:

1. The 'product quality' including 4 core quality dimensions: 1. Performance; 3. Conformance; 4. Durability; 5. Reliability;
2. the 'service quality' including 2 extra dimensions: 2. Features; 6. Serviceability;
3. The 'image & design' including: 7. Aesthetics and 8. Image.

For a distinct group of customers, these 3 aspects of value can be determined in terms of money (a 'fair price', see Annex 5.3), where:

(total value) = ('fair price' of the 'product quality')
+ ('fair price' of the 'service quality') + ('fair price' of the 'image').

The value side of Figure 5.2 depicts the 'fair price' for each aspect in €. The sum of the three fair prices in the EV Wheel equals the total value of the product/service or PSS for the total life cycle.

The three four options to enhance product value are:

- Enhance the functionality of a product during the use phase.
- Extend the use phase.

- Enhance Image and Design
- Change the design to give the product a positive (or less negative) value at the End of Life.

The eco-costs side of Figure 5.2 depicts the eco-costs (in €) of the product/service or PSS for the five aspects of the total Life Cycle:

- Materials.
- Production.
- Distribution.
- Use.
- End of Life.

The EV Wheel is basically a communication tool: it shows in one picture the advantages (and disadvantages) of a specific design in comparison with other designs. The upper part of the wheel shows if, and in which aspects, a product/service or PSS is attractive to customers. The lower part shows if, and in which aspects, the product has a low ecological burden. A sustainable product combines a big shaded area in the upper part and a small shaded area in the lower part. The wheel makes it clear where to focus so as to improve the design.

The EVR of a total product / service or PSS is the sum of the eco-costs of the five aspects (the lower part of the wheel) divided by the sum of the fair prices (the upper part of the wheel).

5.4 Design strategies in the business chain of a product: the case of a 28” CRT TV

How to optimise a production and distribution chain through the use of the EVR model, can be explained by use of a simple example of a 'standard' television with a 28" CRT (data is given for a typical configuration, slightly rounded off, for confidentiality reasons). See Tables 5.1 and 5.2.

	Eco-costs (€)
CRT	35,30
speakers	10,20
enclosure	11,80
chassis	44,30
packaging	3,40
TOTAL	106,00

		ΔValue (€)	Value (€)	Δ Eco-costs (€)	Eco-costs (€)	EVR
1	Components	200	200	106	106	0.53
2	Assembly	150	350	33	139	0.22
3	Distribution	50	400	36	175	0.72
4	Advertising, etc	50	450	10	185	0.20
5	Retail	225	675	23	208	0.10
	TOTAL	675		208		0.31

Table 5.1 (left). Eco-costs of components 28" CRT TV.

Table 5.2 (right). Eco-costs of the production and distribution chain of a 28” TV (tentative).

Note. Table 5.2 excludes the use phase and the End of Life phase (price levels 1999).

Table 5.2 shows that:

- The eco-costs of the components are approximately equal to the eco-costs caused by the assembly, distribution, advertising and retail of the television set. Note that the eco-costs for 'use phase' and 'End of Life' will be dealt with later in the 'low energy' TV case, see Table 5.3.
- The EVR gets lower towards the end of the production and distribution chain. This is because of the higher labour content of service and image, which is added to the product at the end of the production and distribution chain.

Figure 5.3. The value and the eco-costs cumulative along the production and distribution chain.

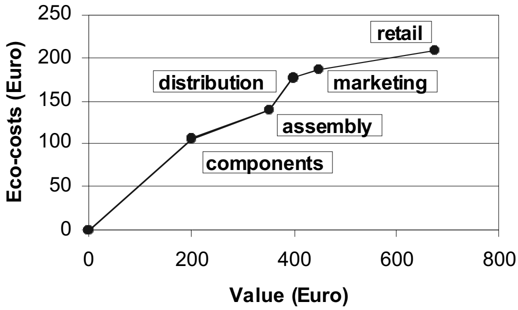


Figure 5.3 depicts the development of the value and the ecocosts in the production and distribution chain. From Figure 5.3 it is clear that the distribution step requires extra attention from the environmental point of view: the increase of the costs (value) is low, but the increase of the eco-costs is relatively high.

The two dimensional approach of the eco-costs / value ratio seems to be crucial in calculating as well as in understanding the elements of the eco-effciency of a product, a service or a PSS. It reveals the fundamental differences between environmental strategies in each step of the chain:

- Making production processes cleaner (lowering the ecocosts at often a constant cost level).
- Environmental material selection (lowering the ecocosts at often a higher cost level).
- Savings in e.g. transport (lowering both costs and eco-costs).
- Improvement of the perceived value (enhancing the value without adding considerable extra eco-costs).

Only a good understanding of value, costs and eco-costs along the total chain can lead to the required improvements in design (Design for Sustainability). It is important here to mention that the EVR method also accommodates a system for quick estimates of the ecocosts (instead of a laborious full LCA), applying tables of eco-cost per kilogram for materials and eco-costs per Joule for energy sources (see: www.ecocostsvalue.com).

With regard to the strategic level of product portfolio management, it is essential to understand the chain[38] in order to manoeuvre a corporation into a position which is suitable for a sustainable future.
A low EVR indicates that the product is fit for use in a future sustainable society. A high EVR indicates that the value/costs ratio of a product might become less than 1 in the future (since stricter governmental regulations on the environment will result in the virtual eco-costs becoming part of the 'internal' coststructure), so there is no market for such a product.

Since the success of a PSS in the market depends whether or not consumers will buy the product, it is important that consumers can understand the concept of eco-costs. The perceived value is what they feel and think about the product, the cost is what they pay for it, but the concept of eco-costs is new for them. This means that companies face a new challenge in the marketing of eco-costs.

5.5 Dilemmas on strategies for marketing and pricing. Case: a 'low energy' television

In the previous sections, only the first steps in the chain have been analysed. In this section we will analyse the total chain, especially the aspects of the 'Use phase' of a product.

Equivalent to the Total Costs of Ownership [TCO] (Life Cycle Costing) approach of financial evaluation, the Total Eco-Costs of Ownership [TECO] is defined and applied to a 28" television. See Table 5.3 for two cases:

- The 'American family' in Europe, watching 6 hours per day on average.
- The 'European young bachelor', watching 1.5 hour per day on average.

[38] For a product or service system, the bundle of business activities is also-called the 'profit pool', rather than the product chain (Gadiesh, 1998).

Table 5.3. The Total Costs of Ownership [TCO] and the Total Eco-Costs of Ownership [TECO] of a standard 28 TV and a 'low energy' 28" TV, for two onsumer market segments.

All prices in 1999	Watching 6 hours per day (average in a year)			
	standard 28" TV		*"low energy" 28" TV*	
	Value (costs)	Eco-costs	Value (costs)	Eco-costs
Purchase	675	208	776 (+15%)	239 (+15%)
Energy 'watching'	174	239	130 (-25%)	179 (-25%)
Energy 'stand by'	13	18	10 (-25%)	13 (-25%)
'End of Life'	30	3	30	3
TCO and TECO	892	468	912 (+2%)	434 (-7%)

All prices in €	Watching 1.5 hour per day (average in a year)			
	standard 28" TV		*"low energy" 28" TV*	
	Value (costs)	Eco-costs	Value (costs)	Eco-costs
Purchase	675	208	776 (+15%)	239 (+15%)
Energy 'watching'	44	60	33 (-25%)	45 (-25%)
Energy 'stand by'	16	22	12 (-25%)	16 (-25%)
'End of Life'	30	3	30	3
TCO and TECO	765	293	840 (+10%)	303 (+3%)

It is assumed that the life time of the television is 10 years, and that the energy consumption is 100 W during watching and 2.5 W during 'stand by' (at a renewable electricity price of 19.60 € per GJ).

The data of Table 5.3 on the standard 28" TV are derived from two confidential studies on the subject: one study for Philips (1997) and one study for Sony (1995). These data are tentative and tend to be different for each specific design.
The data on the low energy TV are of a more hypothetical character: the energy consumption of the standard TV could be reduced by a more advanced system design (there are several options), however; at a higher level of production costs.
Suppose that a new 'low energy' type of television will be launched with the following characteristics

- The price and related ecocosts are 15% higher.
- The energy consumption is 25% lower (for watching as well as stand by).

The first question now is whether this new television is attractive from the environmental point of view. The answer is:

- Yes; when we watch the TV for more than 6 hours per day on average (column 5, Table 5.3).
- No; when we watch for less than 1.5 hours, since the extra eco-costs in the production chain is more than the savings of eco-costs during the use- phase (column 5, Table 5.3).

From governmental point of view it seems to be attractive to stimulate the low energy television by subsidising (or differentiating through taxation), to influence the purchasing decision of the consumer in the retail shop (remember that the purchase price is 15% higher without subsidy). However, when the low energy television is purchased by people who watch less than 1.5 hours per day, the government is *subsidizing the eco-costs*! The only solution is to design a separate marketing strategy for each of the market niches. In all circumstances the issue has to be clearly communicated to the market (the consumers).

The second question is whether or not the government should stimulate the replacement of old televisions by the new low energy televisions. From a governmental point of view this seems to be attractive since it results in less energy consumption. However, the consequence of this action is that in that case televisions are recycled at an earlier (premature) stage than is required from the durability point of view. This dilemma can be resolved by calculating the effects of replacement of the old type television (see Table 5.3):

- Assuming that the life time of a TV is 10 years, the related eco-costs per year of the depreciation of the standard TV is 20.8 € per year.
- The savings of the eco-costs of energy is 65 € over a period of 10 years, so 6.5 € per year.

The conclusion is that, from an environmental point of view, it is not reasonable to throw away the old TV when it is still working: the energy savings are not enough to counterbalance the negative effect of throwing away the old TV before its optimal lifetime.

The question whether or not to invest in a system which has a lower environmental impact, is not only an issue in the market of appliances. It is one of the major issues with regard to investment strategies of industries and governments on the road towards sustainability. The policy with regard to investments in new 'cleaner production' systems is dealt with in the next section.

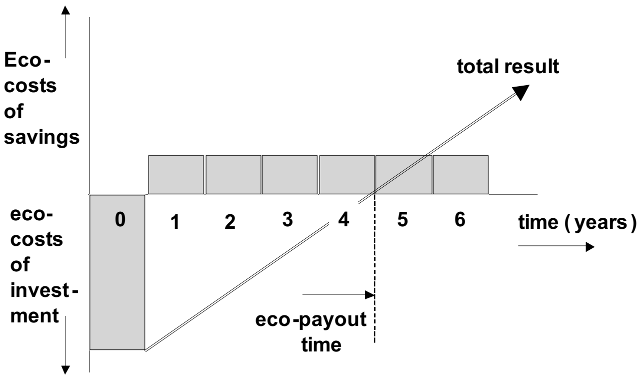
5.6 The ECO-payout time for investments

It is important for companies and governments to know which investment is the best choice on the road to sustainability. Analogous to the 'payout time' as an investment criterion in financial evaluations, the concept of the ecopayout time is introduced:

(5.1) Eco-payout time = Eco-costs of the investment – Eco-costs of the savings

The meaning of formula (5.1) is depicted in Figure 5.4. This figure shows that only after the pay-out time period, the savings are bigger than the expenditure at the start. So after the pay-out time period we have a positive balance on eco-costs.

Figure 5.4. The eco-payout time



In practice, it would be a major step forward towards sustainability if companies not only base their investment decisions on the pay-out time (or other economic criteria), but also consider the eco-payout time. In doing so, these companies will position themselves step by step better in the competitive world of future sustainability, since they will move towards production systems with low eco-costs (towards 'cleaner production').

Formula (5.1) can be transformed to:

(5.2) Eco-payout time = [(value of investment) × $EVR_{investment}$]I
[(value of savings per year) × $EVR_{savings}$]

Or:

(5.3) Eco-payout time = Pay-out time × $EVR_{investment}$ EVR_{saving}

When a company bases the investment decisions on such a double criterion (pay-out time as well as eco-payout time), investments in savings of energy or material will prevail (before, e.g. savings in labour) since the EVR of labour is lower than the EVR of materials and energy.

There are types of investments that enable cost savings which are normally abandoned directly after the depreciation period (= the payout time). In such cases one should be certain that the eco-payout time is shorter than the payout time, otherwise a negative balance of eco-costs is left. In these cases one should fulfil the following criterion:

(5.4) $EVR_{investment} \leq EVR_{savings}$

For investments in computer software for instance, this criterion is no problem: labour is replaced by labour. However, for investments in computer hardware this criterion is a problem: the EVR of a computer is higher than the EVR of the labour it replaces. In this case one should keep the hardware longer than the depreciation period from the environmental viewpoint!

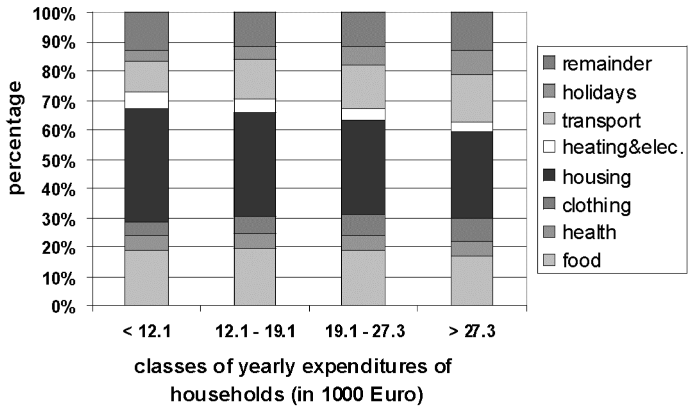
In the case of replacement of production facilities (investments without additional savings), the ecological balance is always negative: one should always consider here the alternatives of maintenance and/or renovation. This also applies for office buildings.

5.7 The EVR model and the buying pattern of consumers. The 'rebound effect'

De-linking of economy and ecology is also related to the lifestyle of consumers. Under the assumption that most households spend what they earn in their life, a low level of EVR of the household expenditure is the key towards sustainability. Then, the 'virtual eco-costs' will be low, even at a high total value of the expenditures. There are two levels for achieving this:

1. At the product level: the delivery of eco-efficient (low EVR) products and services by the industry.
2. At the consumer's level: the change of lifestyle in the direction of low EVR consumption patterns.

At the product level, our society is heading in the right direction: gradually industrial production is achieving higher levels of the value/costs ratio and this means it is becoming 'cleaner'.
At the consumer's level, however, our society is suffering from the fact that the consumers preferences are heading in the wrong direction: towards products and services with an unfavourable EVR (like bigger cars, more kilometres, intercontinental flights for holidays). These unfavourable preferences can be concluded from Figure 5.5.



EVR: transport & holidays = 0.70 – 1.50 ; housing 0,3 ; clothing & health 0,2

Figure 5.5. Preferences o expenditures households the Netherlan (CBS, 1995)

Figure 5.5 shows that people in the Netherlands (and probably in the other EC countries) spend relatively more money on cars and holidays when they have more money available. Other studies (Kramer, 1998) show that people tend to go on intercontinental holidays when they can afford it.

It is obvious that these preferences will become a big problem when people become richer, since the EVR of food, health, clothing and housing is much lower than the EVR of transport and (inter)continental holidays by plane:

- The EVR of food, health, clothing and housing is estimated in the range of 0.3–0.4
- The EVR of transport by car in Europe is estimated in the range of 0.8–1.0.

So, when European households get richer, their spending gradually produces a higher EVR, which is the wrong direction in terms of eco-efficiency and sustainability.

Consumer preferences are relevant for the design of products and services due to the so-called 'rebound effect' (see Figure 5.6). When eco-costs are reduced by 'savings', the economic value (costs to the consumer) is reduced as well, so the consumer will spend the money somewhere else. In the example of Figure 5.6, savings on product type 1 have a positive effect on the eco-costs, if the money is spent on products with a lower eco-costs/value ratio (e.g. product type 2). Example: savings on travel costs are spent on housing. However, savings on product type 2 may result in higher eco-costs if the money is spent on products with a higher EVR (e.g. product type 1). Example: savings on housing costs are spent on travel. The conclusion is that 'savings' are only positive for the environment when savings are achieved in areas with a high EVR.

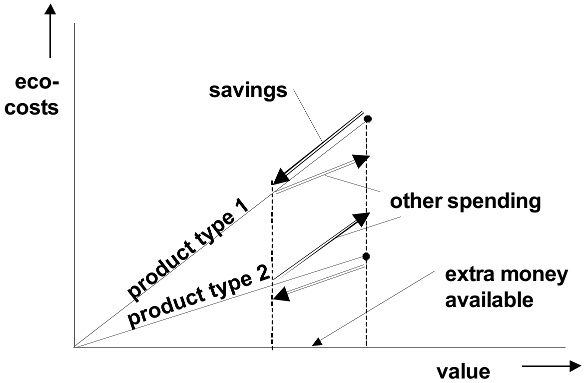


Figure 5.6. The 'rebound effect' of consumer spending.

A typical example of the 'rebound effect' relates to the efficiency increase of light bulbs: when consumers spend the saved energy on more light (e.g. in their gardens) or on electricity for other domestic appliances, this doesn't help much in terms of sustainability.

In general, however; one may conclude that savings on energy can have a positive effect in terms of sustainahility, since the EVR of energy is relatively high [0.7 … 1.2] (Chapter 3) in comparison with other expenditures. Savings on luxury goods (generally a low EVR because of the high labour content: 0.1 … 0.3) might be negative since the 'rebound' might relate to the use of more energy (e.g. in the form of travel).

In product design, savings in energy often require higher product costs. An example of such a saving relates to making cars lighter (in order to reduce fuel consumption). Figure 5.7 shows the result of calculations for middle class German cars and for European fuel prices (Saur, 1999).

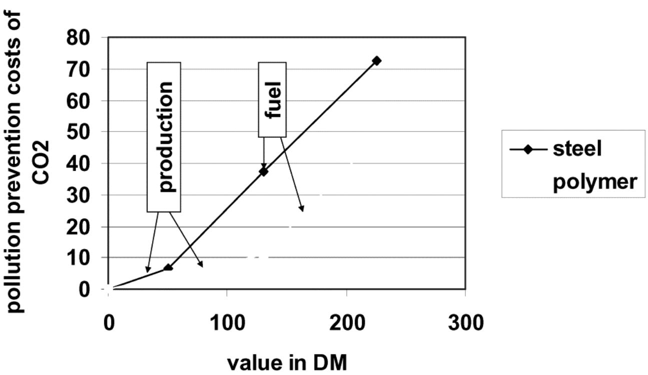


Figure 5.7. The Total Costs of Ownership (TCO) and the pollution prevention costs of CO_2 for two alternative designs for a part of the coach-work of a middle class German car.

The left hand side of the two lines relates to the production (the End-of-Life phase included) of a specific part of the coach-work of the car. The right hand side of the two lines depicts the effect of the fuel consumption on that specific part of the coach-work (0.305 litre fuel per 100 km per 100 kg). At a total life time of 250,000 km, SMC (a polymer) is break even with steel from the economic point of view, but much better from the environmental point of view. This is an example where there is no 'rebound effect', since the economic savings on the fuel equal the extra costs of the coach-work.

It should be mentioned here that the change in cost structure (from energy costs towards product costs) may have a serious impact on the marketing of the product. When the example of coach-work is applied to the total car (a hypothetical case), the price of the car will go up with nearly a factor 2.5, which may only be marketed by means of total lease concepts (for the total life time of the car!).

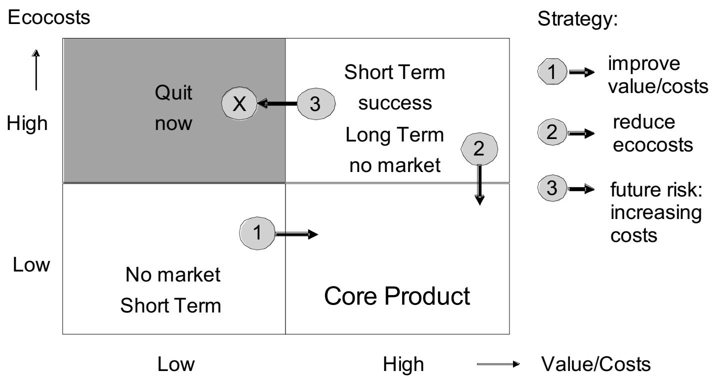
5.8 Conclusions

The development of a sustainable society needs a combined approach:

- By the industry: the delivery of eco-efficient (low EVR) products and services.
- By the consumers: a change in life-style towards low EVR consumption patterns.

Governments should lead industry in the right direction (e.g. by regulations), and should stimulate consumers to make the right choices in terms of the EVR.
With regard to product portfolio management of companies, the EVR model shows the clear implications in the matrix for products and service systems of Figure 5.8.

Figure 5.8. Product portfolio matrix for product strategy of companies.



The basic idea of this product portfolio matrix is the fact that each product or service system is characterized by:

- Its short term market potential: the value/costs ratio.
- Its long term market requirement: low eco-costs.

In terms of product strategy, the matrix results in 4 strategic directions:

1. Enhance the value/costs ratio of a sustainable design with a sound eco-costs level to make it fit for short term introduction in the market.
2. Lower the eco-costs level of current successful products to make it fit for future markets.

3. Abandon products with a low value/costs ratio and high eco-costs as well.

There appears to be four fundamentally different opportunities to enhance the ecocosts/value ratio:

- Making production processes cleaner (1owering the ecocosts at the same value).
- Better materials selection (lowering the eco-costs at often higher cost levels).
- Savings in e.g. transport (lowering both costs and eco-costs).
- Improvement of the perceived value (enhancing the value without adding considerable extra costs).

Understanding these aspects of product development and marketing strategies is essential to manoeuvre a corporation into a position which is fit for a sustainable future. Furthermore, the eco-payout time should play a role in the decision taking process to manoeuvre a company and its products into a better position for the future (e.g. a lower EVR).

Annex 5.1. The virtual eco-costs and the EVR model

The basic idea of the EVR model is to link the 'value chain' (Porter, 1985) to the ecological 'product chain'. In the value chain, the added value (in terms of money) and the added costs are determined for each step of the product 'from cradle to grave'. Similarly, the ecological impact of each step in the product chain is expressed in terms of money as well the so-called eco-costs, see Figure 5.9.

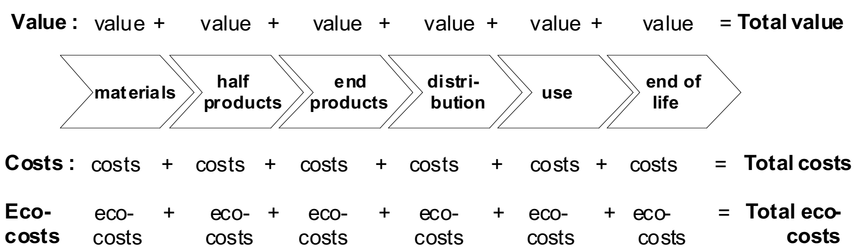


Figure 5.9. The basic idea of combining the economic and ecological chain: 'the EVR chain'.

The eco-costs are virtual costs, resulting from a 'what if' calculation on 'marginal prevention costs': they are a norm for the costs of measures which have to be taken to make (and recycle) a product 'in line with earth's estimated carrying capacity' (see. Figure 5.10). These costs have been determined on the basis of technical measures to prevent pollution and resource depletion to a level which is sufficient to make our society sustainable (Chapter 2 and 3).

Since our society is yet far from sustainable, the eco-costs are virtual: they have been estimated on a 'what if' basis. They are not yet fully integrated in the current costs of the product chain (the current Life Cycle Costs). The ratio of eco-cost and value, the so-called Eco-costs/Value Ratio, EVR, is defined in each step in the chain as:

(5.5) EVR = eco-costs/value

For one step in the production and distribution chain, the eco-costs, the costs and the value[39] are depicted in Figure 5.1.
To determine the eco-costs, five components of the ecocosts have been defined, being 3 'direct' components plus 2 'indirect' components:

- Virtual *pollution prevention* costs, being the costs required to reduce the *emissions* of the production processes to a sustainable level (Chapter 2).
- Eco-costs of *energy,* being the extra price for renewable energy sources.
- *Materials depletion* costs, being (costs of raw materials) × (1 – α), where α is the recycled fraction; for more information see (Chapter 3).
- Eco-costs of *depreciation,* being the eco-costs related with the use of equipment, buildings, etc.
- Eco-costs of *labour* being the eco-costs related to labour, such as commuting and the use of the office (building, heating, lighting, electricity for computers, paper, office products, etc.).

Based on a detailed cost-structure of the product, the eco-costs can be calculated, by multiplying each cost element with its specific eco-costs/value ratio, the EVR. These specific EVRs have been calculated on the basis of LCAs (see the tables at www.ecocostsvalue.com)

Figure 5.10. The definition of eco-efficiency (WBCSD, 1995).

the delivery of
competitively priced goods and services
that satisfy human needs and bring quality of life,
while progressively reducing
ecological impacts and resource intensity
- through the life cycle -
to a level in line with
earth's estimated carrying capacity

- The EVR model has been based on the definition of eco-efficiency, as developed by the WBCSD, see Figure 5.10. The part of this definition above the dotted line is describing the value of a product; the part under the dotted line is defined by the ecocosts.

39 Within the production chain, 'value' equals the sales price. At the end of a production chain, where the consumer buys the product, the way the 'value' has to be assessed is slightly different (Chapter 3): from the consumers point of view the value equals the 'fair price' (Gale, 1990), see Annex 5.3. For a short description of the 'costs-price-value' model see Annex 5.2

Annex 5.2. The costs-price-value model

In elaborating the concept of eco-efficiency as defined by the WBCSD and the basic idea of the EVR model as depicted in Figures 5.1 and 5.9, it is essential to understand the differences between costs, price and value as they are defined in modern management theories (like Total Quality Management and/or Continious Improvement).

The classical management paradigm describing the function of costs, price and value is depicted in Figure 5.11. In the eyes of the producer, profit is a result of the difference between the costs of a product and its price. Managers try to reduce the costs as much as possible and get a price as high as possible.
However managers know that the end user (consumer) will buy the product only when, in his or her eyes, the perceived value is higher that the price.

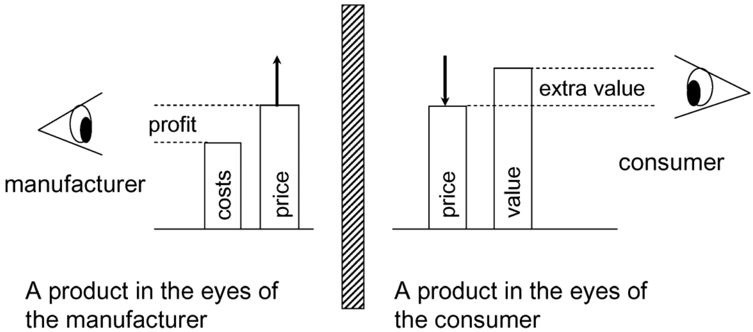


Figure 5.11. The classical paradigm is price driven, which leads to 'cost cutting'.

In the classical management paradigm, the manager has no choice: when the price gets too high, there will be no buyers, so the only thing he can focus on is reducing costs. In this paradigm, measures for environmental protection add costs must be kept to a minimum to cut costs.

In modern management, the strategic focus is on the ratio of value and costs, as is depicted in Figure 5.12. A big difference between value and costs creates a variety of strategic options for setting the right price (more profit by optimisation of margin per product versus sales volume).

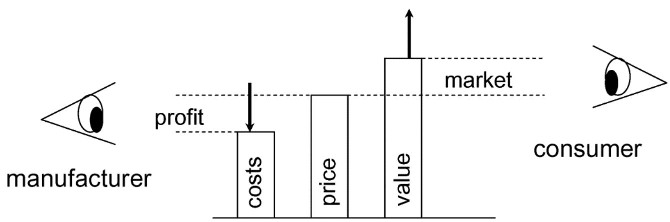


Figure 5.12. The new management paradigm is about enhancing the 'value/costs ratio'.

In classical management, higher value ('quality') always leads to higher costs. In modern management this is not the case: there are many management techniques that lead to a better value/costs ratio. Examples are: logistics (better delivery at lower stock levels), complaint management (satisfied customers with less claims), waste and quality management (less materials better quality). All these examples – there are many more in the field of Total Quality Management and Continuous Improvement – lead to more value at less costs. This is called the *double objective for managers* (Vollmann, 1996) and opens new perspectives to support eco-efficiency (it supports the first part of the eco-efficiency definition of the WBCSD, see also Porter, 1995).
Note that this modern management philosophy is much more than just adding services to existing products. It is about carefully improving the quality of products and services (as perceived by the customer!) by eliminating the non-value adding energy, materials and work.

The question is now whether these modern management techniques always lead to better eco-efficiency. The answer is no (e.g. the use of pesticides in agriculture results in a better value/costs ratio but not to an improved level of environmental protection). That is why the aforementioned definition of eco-efficiency of the WBCSD adds: ".... while progressively reducing ecological impacts…", For this reason, the virtual eco-costs as a single indicator for sustainability has been introduced in the EVR model of Figure 5.1. In this way the 'cost structure' of a product (including services) is linked to the related ecological impacts and material depletion.

Annex 5.3. The dimensions of Quality and the Fair Price

The quality dimensions of Garvin (see Section 5.3) can only be judged by the customers ('as perceived by the customers', measured by customer panels or customer surveys). These quality dimensions can be expressed in terms of the 'fair price' (Gale, 1990).

The technique is that the customer is asked to estimate the value of the total product-service system in terms of the (total) 'fair price'. The fair price is the highest price at which a customer is prepared to buy a product and/or service. When the price of a product is higher than the fair price, the product is considered as too expensive by the customer. When the price is lower than the fair price, the customer considers a purchase as attractive. So the fair price equals the value as depicted in Figures 5.1, 5.11 and 5.12.

In addition to the assessment of the fair price, each quality dimension can be rated:

- In terms of the quality ('value') of each dimension (ranging from 'very poor' to 'excellent').
- In terms of importance of each dimension (ranging from 'not important' to 'very important').

The fair price for the 'product quality', the 'service quality' and the 'image' can be determined then by calculating the weighed averages of the ratings of the quality dimensions, and assigning the corresponding portions of the total fair price to the quality aspects.

An example for an easy, linear, situation is given in Table 5.4. A distinct group of customers (customers within one market niche or one market segment) is asked to assess:

- The total fair price (= total value).
- The score of importance (column (1): "how important is this value aspect for you?" 1 = 'of no importance' to 5 = 'very important).
- The rating of the value (column (2): "what is the quality aspect for you?" 1 = 'very poor' to 5 = 'excellent').

The fair price for the value aspects is calculated then according to the scheme in Table 5.4.

	Value aspect	Importance Score (1)	Value Rating (2)	(3) = (1) x (2)	(4) = (3) / 'total (3)'	'fair price' = (4) x 'total value'
a	Product Q	4	3	12	0.40	360 €
b	Service Q	2	3	6	0.20	180 €
c	Image	3	4	12	0.40	360 €
d	Total			34	1.00	900 €

Table 5.4. Calculation scheme for assessment of the fair price for value aspects (example).

6 Land-use[40]

6.1 Abstract

The environmental impact of land-use can be expressed in terms of a change in biodiversity of flora. We present two models that characterize the negative effects of land-use:

- A model on the basis of species richness.
- A model on the basis of the rarity of ecosystems and their vascular plants.

Each of those models may serve in EIA (Environmental Impact Assessment) of the urban and rural planning of expanding cities, industrial areas, road infrastructure, etcetera.
Moreover, these models might be applied by LCA practitioners to incorporate the aspect of land-use in the environmental assessment of a specific product design.

The results of both models have been applied in practice. Maps of the Netherlands are provided for both models:

- The map based on the rarity of ecosystems differentiates the best of what experts (biologists and ecologists) define as botanical quality of nature; the methodology is operational in the Netherlands and might be applied to other countries as well, however, detailed botanical information is required.
- The map based on species richness has a weaker compliance with the botanical quality of nature, however, the model can more easily be applied to a wider area of the world, since indicative data about species richness is available on a global scale.

The so-called 'eco-costs of land conversion' is proposed as a single indicator, being the marginal costs of prevention (or compensation) of the negative environmental effects on biodiversity caused by change of land-use. These eco-costs of land conversion for the botanical aspects are part of the much broader model of the Eco-costs/Value Ratio, as has been described in Chapter 3.

[40] The original title was: "Characterizing the change of land-use based on flora: Application for EIA and LCA." Published in Virtual Journal of Environmental Sustainability and J. of Cleaner Production (Vogtländer, 2003, Vogtländer, 2004,A).

6.2 Introduction: Land-use, sustainability, EIA and LCA models

6.2.1 The need for a characterization system for land-use in EIA, and its use in LCAs

The increasing use of land for urban areas, industrial areas, road infrastructure, etc., (USDA, 2000) is a major cause of degradation of the biodiversity in our environment (Sala, 2000). In the last decades there has been a growing concern among EIA and LCA practitioners and spatial planning experts about this negative aspect of a growing population and a growing economic wealth (Rees, 2000, Rees, 1996, Jannson, 1994. Van der Ryn, 1986).

It is obvious that, in future planning, the use of land should be optimized as much as possible by making more intensive (e.g. compact) use of cities and industrial areas. In a lot of cases, however, expansion of cities and industrial areas cannot be avoided. In such cases decisions have to be taken on how and where the expansion is planned. This issue is relevant in the field of EIA as well as LCA:

- In the field of EIA and spatial planning of urban and rural areas, it is relevant how to expand built-up areas with a minimal degradation of nature (S. Thorp, 1996; Dutch ministry of LNV, 2000; Dutch ministry of VROM, 2001).
- In the field of LCA, it is relevant to incorporate the negative aspects of land-use in the analyses of product-service systems (Lindeijer, 2000, Weidema, 2001, Köllner, 2000).

The important aspect of the increased use of land is that the environmental impact (degradation) of it depends on where the land-use (or the change in land-use) takes place, since the environmental impact of change of land-use highly depends on its exact location. This applies to a global as well as a local scale:

- On a global scale: the botanical value[41] of tropical rainforests is quite different from that of the deserts.
- On a local scale: the botanical value can be quite different over relatively short distances of a few kilometres, e.g. near dunes, rivers, moors, coastlines, etc.

The local differences of the botanical value of land create a variety of opportunities in spatial planning of densely populated areas: relatively small changes in spatial planning can often save valuable areas of botanical nature. A characterization system of botanical value can guide governments in such cases to find better solutions in a structural way.
The high variety of botanical value on a local scale is an opportunity as well for engineers of new production systems (facilities) of companies: they can influence the

[41] The botanical value of land is defined by a combination of completeness and rarity of the ecosystems.

LCA's of their production systems in a positive way by taking the botanical value of land into account in site selection studies.

For the materials in a LCA, it does make sense to apply a global weighed average of the land-use parameters of mines (in the case of metals), and of forests (in the case of wood from natural rainforests).

6.2.2 Two characterization systems of biodiversity of flora: species richness and rare ecosystems

There are many aspects of land-use in respect to the subject of sustainability (Lindeijer, 2000, Weidemar, 2001, Vogtländer, 2001,A). In this section we will focus on one aspect, i.e. on the botanical value since this is the only type of characterization system which is currently operational (Lindeijer, 2000). We will examine two types of characterization systems for botanical value:

- A coarse system, based on the number of vascular plant species in a certain area: the system of the 'species richness'.
- A more subtle system, based on the relative species richness of ecosystems as well as the diversity and rarity of ecosystems: the system of the 'rare ecosystems'.

For each typical case, one has to make a choice between these systems (otherwise it would lead to double counting of the effect on botanical value). Each system has its pros and its cons, see Table 6.1.
The system on 'rare ecosystems' has been judged by experts on flora. They concluded that 'rare ecosystems' is a better indicator for botanical value than species richness (Witte, 1998).

6.2.3 Creating a single indicator for the LCA, based on conversion or occupation

The LCA methodology provides a structured way to characterize the environmental burden of conversion and occupation of land. The normal route to develop one single indicator for land-use in LCA comprises of two steps:

Step 1. Define a category indicator, based on the characterization model (ISO 14044). For land-use such a category indicator can be expressed in equivalent m^2'
equivalent m^2 = actual m^2 × quality factor
Note that the quality factor is the ratio between the actual quality and the reference quality, and that the equivalent m^2 refers, in our case, to a certain botanical quality of land, the norm for quality.

Step 2. Define a single indicator for LCA by an evaluation (or normalization) step. In this step, the purpose of the analysis becomes important: is the analysis damage based or prevention based and is the impact described in terms of occupation or conversion (change) of land?

The evaluation step of the eco-costs, as described in this chapter, is based on prevention (or compensation) of conversion of land.

Table 6.1. Advantages and disadvantages of the two characterization systems of biodiversity of flora.

The system of the 'species richness'	The system of the 'rare ecosystems'
Advantages:	Advantages:
- It is the most commonly applied characterization system in the world of LCA practitioners	- It only takes species into account that are indicative of the ecosystems considered (e.i. species which are valuable).
- It is relatively simple and straightforward and data is available in many regions; when they are not available, it is feasible to gather indicative data or to predict the main characteristics based on general observations.	- It uses the relative species richness of ecosystems, so that ecosystems that are species-poor by nature do not get a valuation which is too low.
	- It takes the rarity of ecosystems into account (and, indirectly, also the rarity of species that occur in those ecosystems)
Disadvantages:	Disadvantages:
- All species add to the result in a positive sense, including the species that are part of disturbances (for instance: a heather has a higher species richness when it contains a weed).	- It is not yet commonly applied in the world of LCA and EIA practitioners.
- Certain highly valued but nevertheless species-poor ecosystems (bogs, salt marshes, drifting sand dunes, heathlands) get a valuation which is too low.	- The system is more complex and less easy to comprehend (a reasonable level of biological as well as mathematical knowledge is required); moreover, this system requires more detailed information about the flora.
- The system does not account for the fact that some species (especially the rare and threatened ones) are valued higher by nature conservationists than other species.	

Section 6.3 will show how Step 1 is applied to the botanical value of land in terms of species richness or in terms of rare ecosystems.
Section 6.4 will provide a method to arrive at a single indicator for land-use in the LCA (Step 2) is provided.
Section 6.5 gives a short discussion on results of both characterization systems, and gives conclusions with regard to further applications.

6.3 Characterizing the botanical value of land

6.3.1 The characterization system and the category indicator for 'species richness'

Species richness is characterized by the number of species of vascular plants S in a certain area A. It is one of the most applied measures of characterizing the botanical aspects of land in LCAs (Lindeijer, 2000).
For many Western European countries, data on S are available. For the Netherlands, field data are available on a grid of 1 km^2, see Figure 6.1. This map has been derived from FLORBASE. FLORBASE is a database with counted species of vascular plants in

the wild at a national grid of 1 km^2, 35,000 km^2 in total. This database is a compilation of data from the Provinces, land owning organizations, institutes and private persons.
We express land-use in terms of actual m^2 $\times$ quality factor of land before and after the change. The quality factor is defined as the counted total number of vascular plant species, S, divided by the quality norm for it, $S_{ref.}$.
We now introduce the category indicator for species richness of land, SRI (Species Richness Indicator), which is calculated as the area A (m^2) multiplied by the quality factor for it, $S/S_{ref.}$:

$$(6.1) \qquad \text{SRI} = \text{A} \times S/S_{\text{ref}}$$

So SRI is expressed in terms of equivalent m^2 of nature.
Note that S and $S_{ref.}$ have to be both defined for the same area (e.g. the number of species at 1 km^2)
The environmental effect of the change of land-use is described now as

$$(6.2) \qquad \Delta\text{SRI} = A \times \Delta S/S_{\text{ref.}},$$

where Δ denotes the difference of S and SRI before and after the change.

For the quality norm of S in the Netherlands, $S_{ref.}$, a value of 250 vascular plants species for 1 km^2 is proposed. S is more than 250 for 11% of the total area of the Netherlands. Areas above 300 are very scarce (4%), areas above 200 are quite common (25%), see also Figure 6.1.
For Germany, England and the northern part of France, the same quality norm of 250 species for 1 km^2 is proposed, since the species richness in these countries is in the same order of magnitude as in the Netherlands (W. Barthlott, 1998). For species richness of other countries, see Annex 6.2.

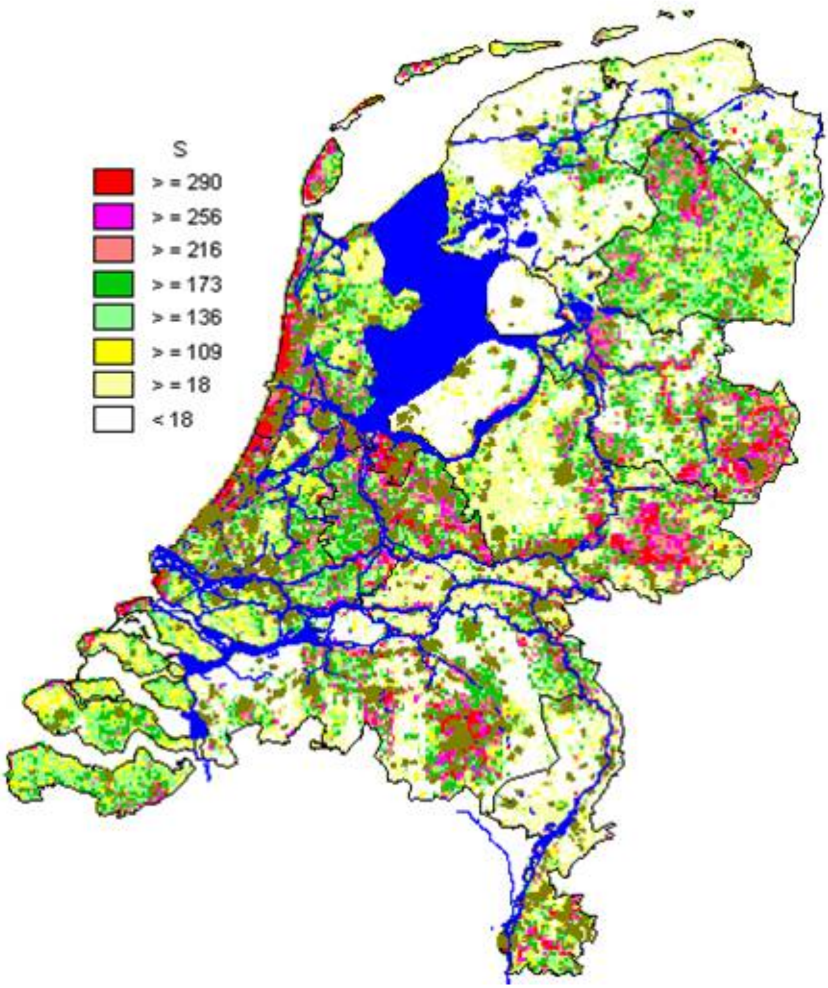


Figure 6.1. The species richness, S (of 1 km^2), in the Netherlands.

Cumulative areas (percentage of total) for S :
> 290, 5%
> 256, 10%
> 215, 20%
> 173, 35%
> 136, 50%
> 109, 60%
> 18, 80%
< 18, 100%

Cities and industrial areas have a brown colour.

6.3.2 Estimation of S in LCA

For EIA applications, *S* must be either available from a database, or must be determined in the field, since detailed data on local level is required for a sound analysis. For LCA purposes, however, the exact location is often not – or not yet – known. In such a case, one might apply a methodology to assess *S* on the basis of the CORINE land classification combined with the size of such an area of land. We propose a calculation system which is in line with the basic approach of the 'Species-pool Effect Potential' of Köllner (Köllner, 2000). The results of such calculations, however, must be regarded as first order estimates, and must be interpreted with great care.
Köllner (Köllner, 2000) analysed data on *S* of various types of land as a function of the size of that type of land, applying the following correlation of *S*:

(6.3a) $S = S_{1\,\mathrm{ha}}(A_{\mathrm{TA}})^{b}$

where:

- $S_{1\ ha}$ is the counted number of species (vascular plants) on 1 hectare (= 0.01 km^2) of a certain type of land
- A_{TA} is the actual size of the total area (hectare) of that type of land
- b is the *species accumulation rate* (the slope of the correlation between the measured value of S and the size of the area on log-log paper)

Table 6.2 provides the data of Köllner (Köllner, 2000) for $S_{1\ ha}$ and b, and subsequent calculations of S for other areas, according to equation (6.3a).

The data of Köllner are based on sample sizes of 1 to 20 ha. Calculation of data by equation (6.3a) is restricted to this range (1 ha < A_{TA} < 20 ha), since further extrapolation is hardly allowed[42] .

For areas smaller than 1 km^2, $S_{ref.}$ in equation (1) and (2) is proposed according to

$$(6.3b) \qquad S_{ref.} = 137.5 \times (A_{TA})^{b}$$

where b = 0.13 (resulting in $S_{ref.}$ = 250 for 1 km^2), see also Table 6.2.

CORINE nr. x)	Land type	S 1 ha	b	Species richness predicted with equ. (3)			
				S 2 ha	S 5 ha	S10 ha	S 20 ha
1.1.1	Continuous urban	10	?	?	?	?	?
1.1.2	Discontinuous urban	55	0.38	72	101	132	172
1.1.3	Urban fallow	90	0.18	102	120	136	154
1.2.1	Industrial area	80	0.22	93	114	133	155
1.2.2.2	Rail area	80	0.22	93	114	133	155
1.2.5	Industrial fallow	105	0.20	121	145	166	191
1.3.4	Mining fallow	85	0.28	103	133	162	197
1.4.1	Green urban	80	0.4	101	138	175	222
1.5	Built-up land	0	0.00	0	0	0	0
2.2.1.1	Conventional arable	10	0.45	14	21	28	39
2.2.1.2	Integrated arable	10	0.50	14	22	32	45
2.2.1.3	Organic arable	25	0.45	34	52	70	96
2.3.1.1	Intensive meadow	15	0.41	20	29	39	53
2.3.1.2	Less intensive meadow	40	0.38	52	74	96	125
2.3.1.3	Organic meadow	45	0.40	59	86	113	149
3.1.1	Broad-leafed forest **)	245	0.13	268	302	330	362
-	Swiss low lands	270	0.13	295	333	364	399
-	$S_{ref.}$ ($S_{ref.}$ = 250 for 1 km^2)	137.5	0.13	150	169	185	203

*) Numbers according to CORINE. Note: ? means that extrapolation is not possible.

Table 6.2. Values for S 1 ha and b according to (Köllner, 2000) S rounded off in units of 5, and predictions of data for S and $S_{ref.}$ of other area sizes by equation (6.3a) and (6.3b).

[42] Above 1 km^2, the value of b will drop drastically to a value between 0.22 (for lower scores of S) and 0.12 (for higher scores of S).

**) Köllner gives $b = 0.36$, but this value results in unrealistic data; $b = 0.13$ is proposed

Example[43]:

When an area of 0.2 km^2 (= 20 ha = 200,000 m^2) is converted totally from 'industrial fallow' to 'industrial area' or to 'intensive meadow', the net effect on SRI in terms of 'equivalent m^2 of nature' can be calculated as follows (applying equation (6.1) and Table 6.2):

- The SRI of the 'industrial fallow' is:
  $A \times S / S_{ref.} = 200{,}000 \times 191 / 250 = 152{,}800$ equiv. m^2
- The SRI of the 'industrial area' is:
  $A \times S / S_{ref.} = 200{,}000 \times 155 / 250 = 112{,}400$ equiv. m^2
- The SRI of the 'intensive meadow' is:
  $A \times S / S_{ref.} = 200{,}000 \times 53 / 250 = 42{,}400$ equiv. m^2

- The net loss of SRI, ΔSRI, of the conversion of 0.2 km^2 'industrial fallow' to 'industrial area' is 40,400 equiv. m^2 of nature.
- The net loss of SRI, ΔSRI of the conversion of 0.2 km^2 'industrial fallow' to 'intensive meadow' is 110,400 equiv. m^2 of nature.

6.3.3 The characterization system and the category indicator for rare ecosystems

Species richness as such, provides only a weak indication of the botanical value (the assumption is: "when there are many species, there is a fair chance that there are valuable species as well"). Therefore the more advanced model of rare ecosystems has been developed, which is described in this section.

The botanical value of a piece of land can best be described by the methodology developed by Witte (Witte, 1998 and Witte, 2000,A). This methodology takes the rarity of ecosystems and their plants into account.

It is a logical step forward to distinguish between species which are important and species which are less important. Witte took 'rarity of the ecosystems' as a main measure of importance. Witte operationalized the methodology by means of the Dutch FLORBASE database.

The basic idea behind this methodology is that every specific ecosystem has its own specific types of vascular plant species. This method distinguishes 28 ecosystem types in the Netherlands.

The methodology results in a score for 'botanical value of one km^2', Q:

(6.4) $$Q = \sum_{i=1,28} V_i C_i$$

[43] This example is a calculation on the so-called 'first order effect'. For more complex situations, where the so-called 'second order effects' play a role, see (Witte, 2000,B). It doubtful, however, whether or not these complex 'second order effects' make sense in practice (Witte, 2000,B).

where:

- V is a parameter for the rarity of an ecosystem type, $1 < V < 10$
- C is a parameter for the relative species richness or 'completeness' (in that km^2) of an ecosystem type in terms of 'indicator species' for that ecosystem type (indicator species are species which occur only in one, two, or maximum 3 ecosystems). $0 < C < 1$.

The summation is to cope with the cases where there are more than one ecosystem within 1 km^2.
The way V and C of equation (6.4) are calculated is briefly described in Annex 6.1.

Witte tested several valuation formulas by showing maps of the province of Utrecht (the Netherlands) to experts in the field of botany. He asked them, single blind (it was not explained to the experts how these maps had been calculated), which map they preferred. Maps based on the species richness S did not score well. The maps based on Q scored the best.

Such a map for Q in the Netherlands is shown in Figure 6.2. In the discussion of Section 6.5 we will show that this map differentiates much better than the map of Figure 6.1.
To arrive at a botanical value in 'equivalent m^2' we propose a national quality norm for Q on the basis of "what is rare (in the Netherlands)?". Such a norm, $Q_{threshold}$, has been determined by a Pareto analysis on all data for the Netherlands (Vogtländer, 2001,A; note that such a norm is basically a political choice):

$Q_{threshold} = 3{,}3$, see the map of Figure 6.2

The botanical value Q is higher than 3.3 for 20% of the total area of The Netherlands.
Equivalent to equation (6.1), the category indicator for botanical rarity of land, ERI (Ecosystems Rarity Indicator), will be expressed in terms of the area, A (m^2), multiplied by ($Q/Q_{threshold}$).

(6.5) $\quad \text{ERI} = A \times (Q \;/\; Q_{threshold})$

So ERI is expressed in terms of 'equivalent m^2 of rare ecosystems'.

The basic idea about the threshold value is that if $Q/Q_{threshold}$ is more than 1, the botanical value of nature is of such an importance that these areas have to be protected (never be converted).

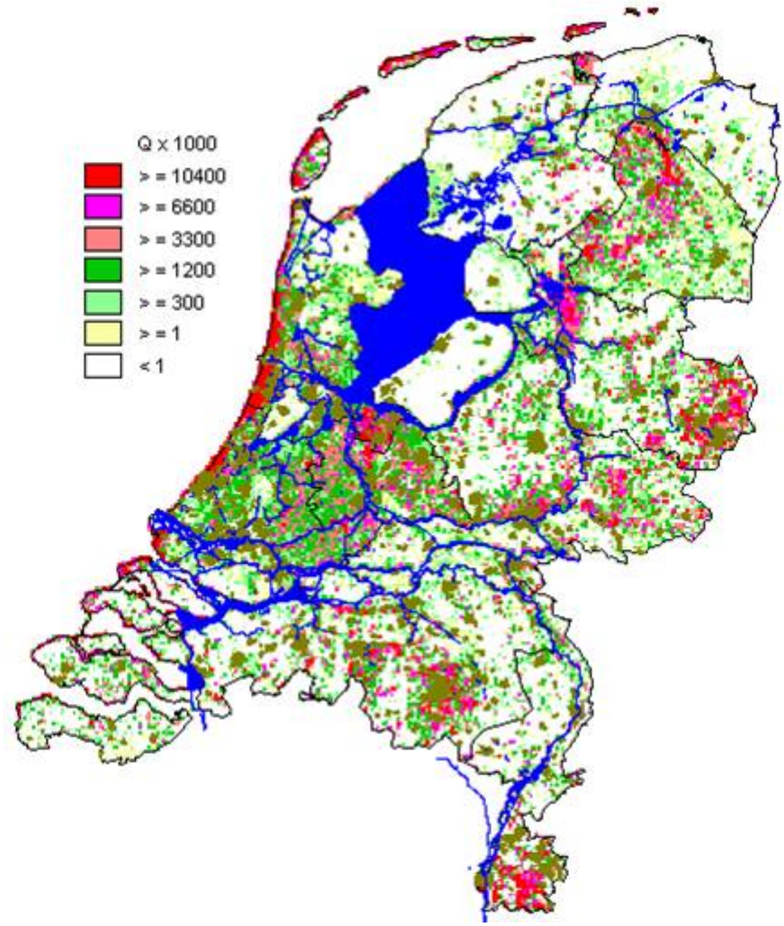


Figure 6.2. The botanical value, Q (of 1 km^2) in the Netherlands

Cumulative areas (percentage of total) for *Q*x1000 :
> 10400, 5%
> 6600, 10%
> 3300, 20%
> 1200, 35%
> 300, 50%
> 1, 60%
< 1, 100% .

Cities and industrial areas have a brown colour.

6.4 A single indicator for land-use in the LCA

6.4.1 Land conversion as a basis for evaluation

The physical impact of land-use can be described in terms of:

- Conversion (change, transformation) of the use of land (with the dimension of m^2).
- Occupation of land for a certain activity (per m^2 and per year).

These two aspects of land use have both been proposed by the SETAC Working Group on Impact Assessment in 1996, see also (Lindeijer, 2000). In formula:

(6.6) land conversion impacts = area $A \times$ Quality difference

(6.7) land occupation impacts = area A × time t × Quality

Both equations can be combined with equations (6.1) and (6.5), since SRI and ERI denote the Quality (in terms of equivalent m^2 of botanical value).
Occupation addresses the impacts of using land, whereas conversion focuses on the impacts of changing the use of land. As the impacts of conversion are considered more relevant than those of occupation (Sala, 2000), we will focus here on the conversion impacts.

The basic idea of conversion of land, with the dimension of (m^2), is that the 'quality of land' is deteriorated when people start to use land. Nature is being destroyed and the environment is degraded at the moment urban and industrial areas or railway and road infrastructure is expanded ('greenfields' are becoming 'brownfields'). The conversion of land causes depletion of scarce 'nature', similar to the resource depletion of materials when virgin materials are used for products.

When a new production facility is planned in a renovated building, the consequence of the conversion approach is that there is no conversion of land (land is re-used, or 'recycled', similar to the approach of the recycling of materials). So the conversion approach in LCA is stimulating the use of existing facilities, instead of creating new facilities on new land. The conversion approach is focussed on the prevention of the expansion of industrial and urban areas: when all economic activities stay within the existing boundaries, there is no impact in terms of conversion.

Land-use described in terms of conversion of land is appropriate to analyse design alternatives for governments (spatial planning) and manufacturing companies (site selection).

6.4.2 The eco-costs for species richness and rare ecosystems

The conversion of a category indicator into the single indicator of the EVR model, the eco-costs, is based on either the prevention costs or the compensation costs of degradation of nature. Compensation means here that somewhere else an extra area of protected nature will be created.
For $(Q/Q_{\text{threshold}}) < 1$, compensation is regarded as feasible as well as realistic.
For $(Q/Q_{\text{threshold}}) > 1$, compensation is not a realistic option (in such a case the 'rare ecosystems' is of exceptional botanical value), so conversion is forbidden (i.e. conversion must be prevented).

The costs related to the creation of a protected nature area (i.e. the compensation costs) are estimated at:

- 4.1 € per m^2 to buy the land (price of agricultural land in the Western part of Europe in 2007),

- 0.6 € per m^2 for the conversion costs (F.J. Sijtsma, 1995), price level 2007.

The resulting total costs of compensation, the eco-costs of species richness as well as the eco-costs of rare ecosystems, are:

4.7 € per equivalent m^2 of nature

In formula:

(6.8a) eco-costs of species richness = $\Delta SRI \times 4.7$ (€) for $(Q/Q_{threshold}) < 1$

Or,

(6.8b) eco-costs of rare ecosystems = $\Delta ERI \times 4.7$ (€) for $(Q/Q_{threshold}) < 1$

Where Δ SRI (or Δ ERI) is the difference between SRI (respectively ERI) before and after the conversion.
Equation (6.8) might be applied to the Netherlands, Belgium, Germany, England and the northern part of France, see (W. Barthlott, 1998). For other countries in the world, a preliminary calculation method of the eco-costs is proposed in Annex 6.2.

6.5 Evaluation

6.5.1 Discussion

The issue of land-use in terms of conversion of land is often a complex problem of contradicting interests of the stakeholders who are involved. The current policy in most western countries is to empower local authorities to meet the needs of their communities. The result, however, is a highly fragmented structure of decision making, with little or no co-ordination to consider regional and long term consequences (Thorp, 1996). 'Not in my back yard' (NIMBY) discussions do often prevail.
In such a situation there is a need for a structured approach in EIA of the land-use conversion.

To quantify the ecological effect of land-use, two aspects play a role:
- The area (km^2) of the land which is to be converted.
- The quality of the land of that area.

Biodiversity is one of the important aspects of that quality.
It is obvious that biodiversity is about the diversity of flora as well as fauna. Vascular plant diversity is an obvious proxi upon which to base a practical indicator of the biodiversity of flora, since vascular plants play a key role in ecosystem functions.

Moreover, diversity and completeness of ecosystems seems to be reasonably well correlated with species diversity (including fauna) in general (Weidema, 2001, Barthlott, 1998).
One may conclude that both methods (the species richness model as well as the rare ecosystem model) can provide a proxi for biodiversity in EIA and LCA calculations on land-use.
Note that the methods can also be applied to EIA calculations with regard to compensation measures (losses in one area might be compensated by creation of a natural area elsewhere).

Comparison of Figure 6.1 and Figure 6.2 reveals that there seems to be a correlation of the data of the two methods on a regional level ("when the species richness *S* is high, the botanical value *Q* is also high in most of the cases").

However, when those maps are analysed in detail on a local level, differences between the two methods can be very significant.
Details are given here for the northern part of the Netherlands: the islands above the Waddenzee (Terschelling, Ameland, Schiermonnikoog, Rottemerplaat and Rottemeroog). See Figures 6.3 and 6.4. It is evident that parts of these islands score low on the species richness map (Figure 6.3), but high on the rare ecosystems map (botanical value, Figure 6.4). An example is the island of Rottemerplaat (Figure 6.5), a protected area because of its high quality of nature: *Q* is higher than 10.4, however, *S* is only between 136 and 109 species at one km^2.

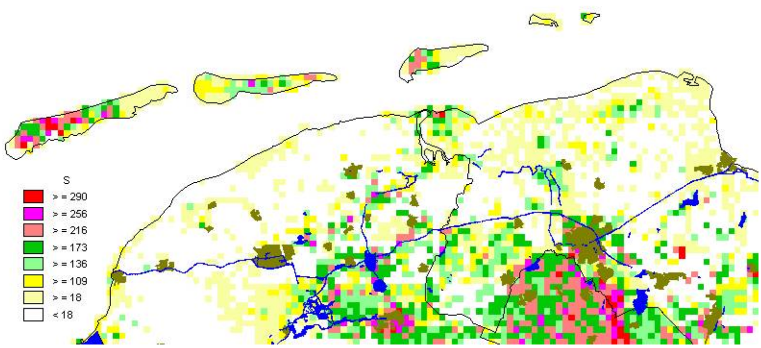


Figure 6.3. The species richness, *S*, in the northern part of The Netherlands.

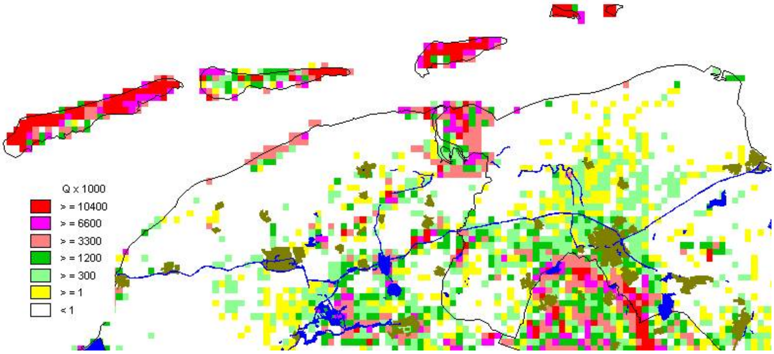


Figure 6.4. The botanical value, *Q*, in the northern part of the Netherlands.

Figure 6.5. The island of Rottermerplaat.

Therefore, local assessments of change of land-use in EIA must always be done by means of the method of the rare ecosystems.
For regional assessments, however, the model of the species richness is accurate enough to provide data with regard to alternative solutions in spatial planning.
So both methods can be valuable tools in the decision making process of EIA.

In most LCA studies, local details are not known, so data on 'species richness' are fair enough as a proxi for the botanical aspects of land-use.

6.6 Conclusions

With regard to EIA and the rare ecosystems model:

- The rare ecosystems indicator is more adequate for EIA than the species richness indicator, since the rare ecosystems model provides better data to base decisions on with regard to spatial planning on a local scale
- Obtaining the detailed data on the flora which is required for the rare ecosystems model, might be a problem in some parts of the world. The complexity of the calculation method might also be regarded as a disadvantage of the model.

With regard to EIA and the species richness model:

- For EIA on the scale of big regions (i.e. more than 5000 km^2), the species richness model might be a practical choice for making comparisons between alternative solutions,
- Data required for the species richness model is available in most of the situations.

With regard to LCA:

- In LCA, the species richness model seems to be accurate enough as a proxi for biodiversity in most of the cases. Even equations (6.3a) and (6.3b) can be used to make an estimate of *S* (note that there are hardly any situations in EIA, where equations (6.3a) and (6.3b) can be considered as acceptable in terms of accuracy).

Annex 6.1. Calculation of the botanical value, Q

The way *V*, *C,* and *Q* of equation (6.4) are calculated is briefly described as follows (see (Witte, 1998) for details).

'Ecosystem types' have been defined based on the following 4 abiotic parameters:

- Salinity (classes: fresh, brackish, and saline).
- Moisture regime (classes: water, wet, moist, dry).
- Nutrient availability (classes: low, moderate, moderate to high and high).
- Acidity (classes: acid, neutral, and alkaline).

Combination of these parameters resulted in 28 ecosystem types relevant for the total Dutch area.

Groups of indicator species have been defined for each ecosystem type. The indicator value, *v*, has been determined to describe the ecosystem – species relationship ($v = 1$ if a vascular plant occurs only in one ecosystem, $v = 1/2$ if a plant occurs in one other ecosystem as well, $v = 1/3$ if a plant occurs in 3 ecosystems, $v = 0$ if a plant occurs in more than 3 ecosystems).

The indicator values *v* have been added up for all *m* species (vascular plants) in a km^2 belonging to a certain ecosystem, resulting in the 'indicator value score', *R*:

$$R = \Sigma\, v_m$$

So *R* is a weighed number of indicator species in one km^2.

The maximum *R* in the database was determined for each of the 28 ecosystems, being the value which is not surpassed in 99.8% (!) of the km-squares (to rule out extremities): $R_{0.2}$ (being the maximum practical weighed count of indicator species in one km^2)

The parameter for the completeness, *C*, has been calculated for each km^2 in the database via a quite complex procedure. This calculation, however, can be approximated within 10% by:

$C = 1$ for $R / R_{0.2} > 0/72$ (the km-square is 'saturated' with indicator species or, in other words, its completeness is very high when *R* is more than 72% of $R_{0.2}$)

$C = 0$ for $R / R_{0.2} < 0.43$ (this threshold determines "… whether an ecosystem may be said to be really present in a km^2 , instead of classifying its occurrence as 'noise' …")

$C = (R - 0.43\, R_{0.2}) / (0.29\, R_{0.2})$ for $0.43 < R/R_{0.2} < 0.72$

The linear range of C might look rather small, but when the results of the scores of C are drawn on maps of The Netherlands, the results appear to be surprisingly good in terms of relevant botanical information.

Species in an abundant ecosystem type are less rare than species in a rare ecosystem type. Therefore, the second main parameter in the model, V, copes with the rarity of an 'ecosystem type'.

V is a function of the occurrence of the ecosystem type in terms of the total weighed area AW (km^2) of that ecosystem type: $AW = \Sigma\ (C \times \Delta A)$ where ΔA is 1 km^2 and the summation is for the total area of the Netherlands

$V = (AW_{max}/AW)^{0.63}$ where AW_{max} is the occurrence $\Sigma\ (C \times \Delta A)$ of the most abundant ecosystem.

Note. V has been slightly corrected for international rare ecosystems. V ranges in the database from 1 to 9.8.

Finally, the importance of land, Q ('Botanical Value of 1 km^2'), can be calculated according equation (6.4):

$$Q = \Sigma\ (V \times C)$$

Note. the summation is used when there is more than one ecosystem type within one km^2.

Annex 6.2. Species richness of vascular plants on a global scale

Figure 6.1 provides the data on species richness on the national level of the Netherlands. For other European countries between 46° and 57° latitude, Table 6.2 or local data might be applied, but what do we do in other areas of the world?

Such a question is particularly of interest for the issues of land-use (land conversion) related to mining of minerals and fossil fuels and to production of wood from rainforests.

In literature, the species richness of vascular plants is provided on the basis of the number of species per 10,000 km^2. The number of species for the Netherlands are in the range of 1000 – 1500 per 10,000 km^2.

To make a preliminary estimate of the eco-costs of land conversion in other parts of the world, the following assumptions have been made:

- $S_{10,000\ km2} = 5 \times S_{1\ km2}$ ($S_{1\ km2} = 250$ for 1 km^2 corresponds with $S_{10,000\ km2} = 1250$ for 10,000 km^2).
- The quality norm of the Netherlands, $S_{1\ km\ nature} = 250$ in equation (6.3), is applied to the other areas as well (so there is one quality level of S for the whole world)

- The eco-costs of species richness of 4.7 € per equivalent m² of nature, see Section 6.4, is applied to the other areas as well (that means that all areas in the world are valued at the same level, regardless of local conditions for marginal prevention or compensation costs).
- The local number of species S before the conversion is the average number of S for the area.
- $S = 0$ after the conversion (for mining as well as conversion of rainforests to argricultural land).

Applying equations (6.3) and (6.4) under the aforementioned conditions results in:

(6.9) eco-costs of species richness $= A \times 4.7 \times S_{10.000\ km2} \ / \ (250 \times 5)$ (€)

where A is the area which is disturbed by the conversion, and $S_{10{,}000\ km2}$ is from Figure 6.6. See also www.ecocostsvalue.com for a full scale picture[44].

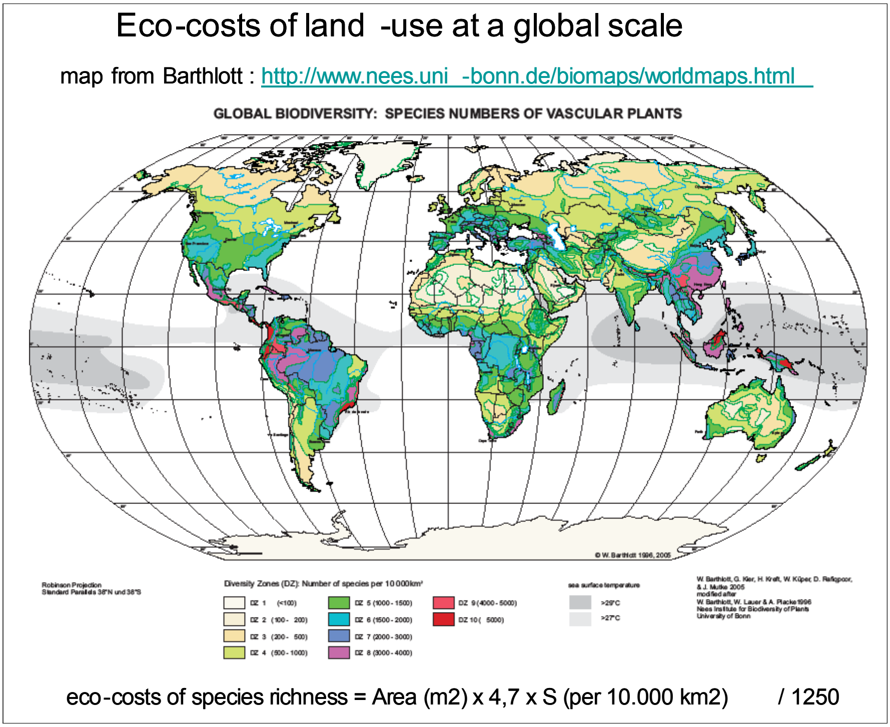


Figure 6.6. Eco-costs of land-use on a global scale.

Rainforests in Panama, Bolivia, Peru and indonesia have a very high species richness: $S_{10{,}000\ km2} = 5000$. The eco-costs 2007 of species richness are for these areas 18.8 € per m².

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[44] Note that Barthlott has updated his data in 2005.

The eco-costs 2007 of rainforest in e.g. Brazil, Venezuela, and Vietnam are slightly lower: 15.0 € per m^2.

Annex 6.3. Other characterization systems for conversion of land in the EVR model

For land-use within the EVR model, the following four main 'endpoint categories'[45] have been proposed with regard to sustainability (subjects which need to be protected, Vogtländer, 2001,A):

- 'Habitat for plants, animals and other species' with biodiversity as an important "impact category', with either species richness or rare vegetation as the most important indicators (indicators for fauna are still under development).
- 'Local habitat for the human being' with 'scenic beauty' as the main 'impact category'.
- 'Food and energy production' with 'net biomass production' as the main 'impact category'.
- 'H_2O cycle function' with 'filter capacity', 'water storage' and 'desiccation' as the main 'impact categories'.

This is depicted in Figure 6.7.

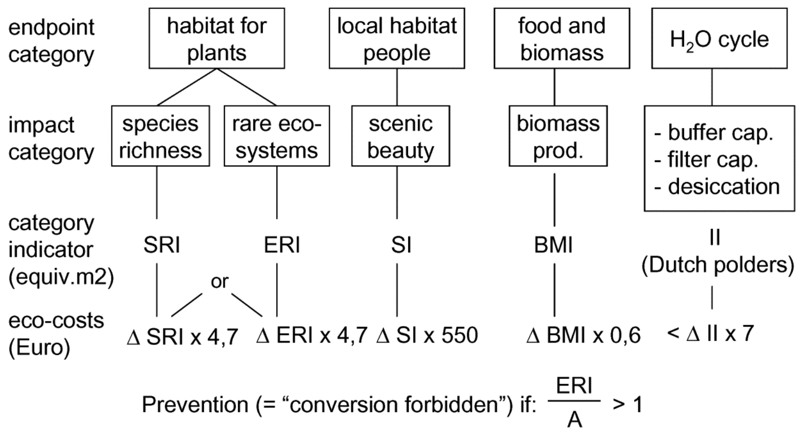

Figure 6.7. Characterization system for land conversion, and the corresponding eco-costs.

The endpoint category 'habitat for plants' (the subject of this section) is basically a proxi for 'habitat for flora and fauna'. The basic idea is that, when vascular plants are safeguarded, the fauna will be safeguarded as well (Weidema, 2001).
The endpoint category 'local habitat for people' is regarded as one of the important sustainability issues (our world is not only for flora and fauna, but for human beings as

[45] The idea of 'endpoint category' has been introduced in the LCA methodology to define the major areas of protection with respect to sustainability (Udo de Haes, 1999).

well). The impact category of 'scenic beauty' is related to urban and rural planning. The basic idea is that the human being has the fundamental right to experience the pleasure of 'nature' (natural beauty, scenic beauty of landscapes and other visual and recreational aspects of parks and landscapes). The 'greenfields' have to be planned not far from the cities, since the need for travelling over long distances has to be kept to a minimum . Scenic beauty is considered as an elementary aspect of the human welfare, it should be protected, and it is therefore a sustainability issue. For details and data of this method (Vogtländer, 2001,A).
Another aspect of 'local habitat for people' is related to the 'eco-costs of noise' of the EVR model. For details and data of this method, see Vogtländer, 2001,A.

The endpoint category 'food and energy production' has had a lot of attention in literature, with biomass production as the main impact category (Lindeijer, 2000).

The endpoint category 'H_2O cycle function' is complex and divers (McKinney, 1998). It seems especially important for spatial planning issues. For the specific situation of Dutch polders, a characterization system has been developed (Vogtländer, 2001,A).
The main functions of land in the H_2O cycle are the filtering (cleansing) function and the storage function of fresh water. Unfortunately, the importance of these functions were neglected for decades. Only recently is there sufficient awareness that these functions are real sustainability issues, and that 'water management' is an indispensable activity for the future.

7 Communication[46]

7.1 Abstract

At Delft University of Technology, a new model has been developed to describe the sustainability of products, the 'EVR model'. This model comprises two concepts:

- The 'virtual eco-costs' as a LCA-based single indicator for environmental impact.
- The EVR (Eco-costs/Value Ratio) as an indicator for eco-efficiency.

In this publication, an experiment is described to test whether the EVR model leads to a good understanding of the eco-efficiency of a product-service combination. In this experiment three separate groups of 8 to 11 people were asked to rank four alternative solutions of a product-service system (the after sales service and the maintenance service of an induction plate cooker) both in terms of sustainability and of general preference. The three respective groups were:

- Customers (among whom representatives of consumer organizations).
- Business representatives from the manufacturing company of the induction plate cookers.
- Governmental representatives (employees of the Dutch ministries of environmental affairs and economic affairs, and of the Dutch provinces as well as consultants involved in governmental policies), all experts in the field of sustainability.

The basic idea was to ask each group to rank the four alternatives after three levels of information input:

Level 1: A basic explanation of the four alternatives. Some major features and characteristics such as price were given, but no environmental data.

Level 2: An explanation of an LCA of the four alternatives, given in 9 impact classes and the Eco-indicator 95.

Level 3: An explanation of the EVR model and the EVR data of the four alternatives.

Each time the group was asked to rank the proposed alternatives in terms of expected environmental performance and of 'best choice in general' ("Which system would you have bought in a real life situation?").

[46] The original title was: "Communicating the eco-efficiency of products and services by means of the Eco-costs/Value Model." Published in J. of Cleaner Production (Vogtländer, Bijma, Brezet, 2002).

From the experiments it can be concluded that:

- The concept of eco-costs was accepted by the majority of the non-experts; they based their ranking on it, and they preferred it rather than direct LCA output or the damage based Eco-indicator '95 data.
- The environmental experts in the governmental group did not directly accept the concept of eco-costs model (they wanted in depth information first); they tended to stick to their existing knowledge of LCA data and the Eco-indicator 95.
- 'Overall' preferences of the customers and business representatives were primarily ranked on the 'perceived value'/costs ratio of the product-service combination; the sustainability of the product-service combination played a secondary role.

7.2 Introduction

In moving towards a sustainable society, three stakeholder groups have a major role:

- Consumers/citizens, who must shift their expenditures towards products with a low environmental burden.
- Companies, which must create product-service combinations with a low environmental burden.
- Governments, which must create regulations and new systems for tax and subsidies that support the required transition.

One of the major issues is that of communication between these three stakeholders. A good interaction between stakeholder groups requires good communication on the subject, which requires all stakeholders to 'speak the same language'.
Currently there seems to be a communication gap between environmental specialists and non-specialists (the majority of the stakeholders). Environmental specialists regularly try to make the situation clear by showing the results of LCAs and the environmental impacts (in terms of damage) of products and processes. However results of an LCA are complex and hard to understand (environmental specialists tend to stress that as well). Many discussions in science about impacts, the complexity of the calculations, and problems with setting priorities, make stakeholders aware of the imminent problems, but do not make clear how to tackle the problem (J. Moisander, 1999, L. Uusitalo, 1999, M. Nas, 2000, L. Steg, 1999).
In terms of providing data on the results of an LCA, there seem to be the following options:

- The full LCA data, which satisfies the LCA specialists, but which is too complex for designers and business managers (so they cannot base there decisions on such data sets).
- The result of one class of emissions only (e.g. in kg CO_2 equivalent), which is clear to designers and business managers, but which is unsatisfactory since environmental problems can be obscured by redesigns which shift the problem towards other classes of pollution.

- One single indicator in terms of 'points' for emissions and materials depletion (e.g. eco-indicator '95 and '99[47], which is clear to designers, but which is opposed by many LCA specialists[48] because of difficulties in weighing the different kind of damage types (classes).
- One single indicator in terms of money (e.g. the EPS indicator), which appeals to business managers, but which is opposed by many LCA specialists.

At the Delft University of Technology, a model has been developed to assess the so-called *eco-efficiency* of products and services[49]. This model is based on the Life Cycle Analysis (LCA) methodology as defined in the ISO 14040 and 14044, and is called the Eco-costs/Value Ratio (EVR) model. It is a decision support tool for designers of sustainable product-service combinations and for business managers to support product portfolio management and marketing strategies.

Right from the start of the development of this new model, it was felt that a model based on prevention costs (instead of the existing damage based models for single indicators) would have good prospects for communication, but that had to be tested first.
Therefore, it was decided to test the way each of the stakeholders (consumers, business managers and governmental representatives) use information to make their decisions. An experiment was designed to find out what kind of preferences prevail in terms of data on the results of LCAs, and how these data influence the final choice (to buy a product-service combination in a real life situation).
During the experiment, the focus was primarily on how the participants made their choices, and on what information they would base those choices.
The participants were given the impression that the aim of the experiment was to make the right selection out of four alternative solutions for a service function (the maintenance of an induction plate cooker), as they would have done in a real life situation. In reality, however, it was not so important what their choice was, but how they made their choice and based on what LCA data.

The primary aim of the experiment was to find out which LCA data set was preferred by the participants: data on the 9 classes of emissions (e.g. kg CO_2 equivalent), the eco-indicator '95, or the eco-costs from the EVR model.

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47 The experiment was held in June 1999. At that time the eco-indicator '99 had not yet been published. It is the impression, however, that the outcome of the experiment would not have been significantly different when the eco-indicator '99 had been applied instead of the eco-indicator '95.

48 In the period before 2000, environmental specialists generally opposed to the idea of a single indicator as such, after 2000 environmentalists got used to it.

49 In 1995, the World Business Council for Sustainable Development (www.wbcsd.org) described the role for industry in their definition of eco-efficiency as:
"*the delivery of competitively priced goods and services that satisfy human needs and bring quality of life while progressively reducing ecological impacts and resource intensity, throughout the life cycle, to a level at least in line with the earth's estimated carrying capacity.*"

The secundary aim was to find out whether, and how, the final purchase decision was influenced by the environmental data.

7.3 The Eco-costs/Value Ratio model

7.3.1 The eco-costs

The basic idea of the EVR (Eco-costs/Value Ratio) model is to link the 'value chain' (Porter, 1985) to the 'ecological product chain'. In the value chain, the added value (in terms of money) and the added costs are determined for each step of the product ´from cradle to grave´. Similarly, the ecological burden of each step in the product chain is expressed in terms of money, the so-called virtual eco-costs '99 (in short eco-costs). See Figure 7.1.

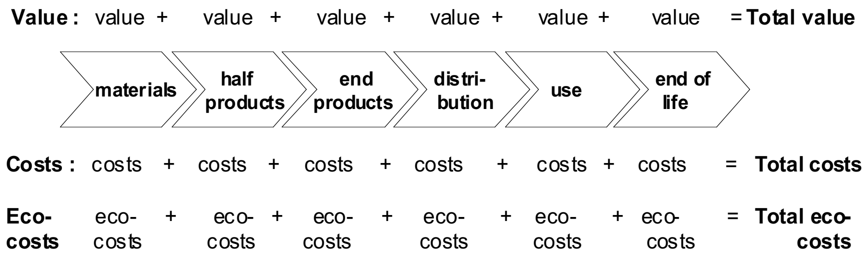

Figure 7.1. The basic idea of combining the economic and ecological chain: ´the EVR chain´.

The eco-costs are 'virtual' costs: these costs are related to measures which have to be taken to make, use and recycle a product "in line with the earth's estimated carrying capacity"(see Annex 5.1). These costs have been estimated on the basis of technical measures to prevent pollution and depletion of materials to a level which is sufficient to make our society sustainable.

Since our society is yet far from sustainable, the eco-costs are 'virtual': they have been estimated on a 'what if' basis. They are not yet fully integrated in the current costs of the product chain (the current Life Cycle Costs)[50].

The ratio of eco-cost and value[51], the so-called Eco-costs/Value Ratio, EVR, can be defined for each step in the chain. For the total life cycle as well as for a part of the chain, the eco-costs, the costs and the value can be calculated, as depicted in Figure 7.2.

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50 The concept of the 'virtual eco-costs' is slightly different from the concept of the 'external costs'. The external costs are related to damage to our environment. The virtual eco-costs are related to the ('marginal') prevention costs, which are required to bring our economy into a state which is sustainable. What both type of costs have in common, is that they are not incorporated in the current economic costs of products and services.

51 Within the business chain, the value equals the market price. Note that the cost for the buyer equals the value for the seller in the business chain. The situation in the Use phase and in the End of Life phase is slightly different. From the consumers point of view the value equals the 'fair price' (Gale, 1994), which reflects the perceived benefit after the purchase. In the End of Life phase the value might be negative.

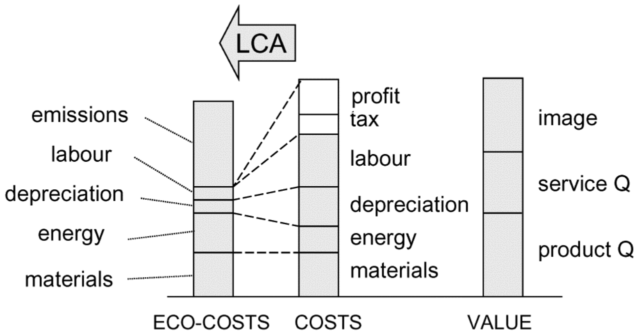


Figure 7.2. The decomposition of ´virtual eco-costs´, costs and value of a product.

The five components of the eco-costs have been defined as 3 'direct' components plus 2 'indirect' components, see Table 7.1. Based on the detailed cost structure of the product, the eco-costs can be calculated for each cost element, applying the LCA methodology as defined in ISO 14040 and 14044.
A detailed description on the way this is to be done is given in (Chapters 2 and 3,).

Table 7.1. The main 5 components of the eco-costs.

Direct components:	Indirect components:
1. Vrtual pollution prevention costs, being the costs required to reduce the *emissions* of the production processes to a sustainable level (Chapter 2)	1. Eco-costs of *depreciation*, being the eco-costs related to the use of equipment, buildings, etc.
2. Eco-costs of *energy*, being the price for renewable energy sources.	2. Eco-costs of *labour*, being the eco-costs related to labour, such as commuting and the use of the office (building, heating, lighting, electricity for computers, paper, office products, etc.).
3.Materials depletion costs being costs of raw materials x (1- α), where α is the recycled fraction[52].	

One of the main elements of the eco-costs are the 'pollution prevention costs' (direct component 1). How these costs are calculated will be described in the next section.

7.3.2 Calculation of the pollution prevention costs

The pollution prevention costs are to be calculated in four steps:
LCA calculation according to the current ISO standards.
Classification of the emissions in 7 classes of pollution (acidification, eutrophication, heavy metals, carcinogens, summer smog, fine dust, global warming).

[52] In theory, one must apply here the 'present market value' (discounted) of the 'sustainable alternative in the future' for the material which is depleted, according to the model of Hotelling (Pearce, 1990). For most of the materials, however, there is no reason to believe that this 'present discounted market value of the sustainable future alternative' deviates much from the current average material prices (examples: tin, copper, iron), since the functionality of these materials can be replaced by alternatives which are not more expensive for their specific functions. So the present price levels can be applied for 'costs of raw materials' in this formula.
An exception is oil as a source for plastics. In the model, the costs for ethanol from biomass has been taken for the 'costs of raw materials' for plastics.

Characterization according to characterization multipliers for Tables of IPPC 2007, CLM-2 and Impact 2002+) resulting in "equivalent kilograms" per class of pollution.

Multiplication of the data of step 3 with the 'prevention costs at the norm', being the marginal costs per kilogram of bringing back the pollution to a level 'in line with earth's carrying capacity'.

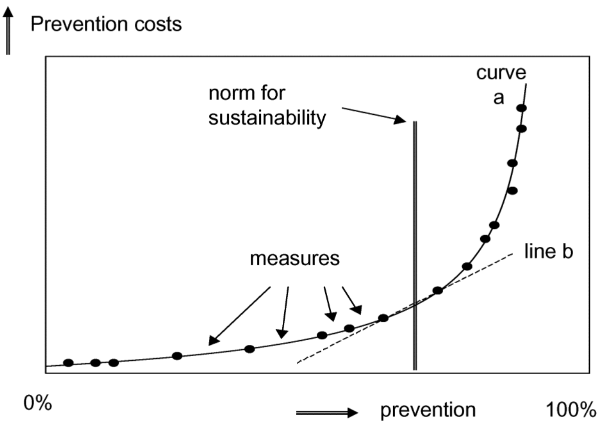
The following 'prevention costs at the norm' have been calculated for the Netherlands and Europe:

- Prevention of acidification 7.55 €/kg SO_x equivalent
- Prevention of eutrophication 3.60 €/kg phosphate equivalent
- Prevention of ecotoxicity (heavy metals) 802 €/kg Zn equivalent
- Prevention of carcinogens 33 €/kg PAH equivalent
- Prevention of summer smog 8.90 €/kg C_2H_4 equivalent
- Prevention of fine dust (winter smog) 27.44 €/kg fine dust PM2.5
- Prevention of global warming 0.135 €/kg CO_2 equivalent.

These 'prevention costs at the norm' are based on the so-called *marginal prevention costs* of emissions. The way these marginal prevention costs are determined is depicted in Figure 7.3. For each type of emission, the costs and the effects (in terms of less emissions) are accumulated for several prevention measures to be taken (a 'what if' calculation). At a certain point of the curve, the *norm for sustainability* is reached. The marginal prevention costs are defined by the costs per kg reduction of the 'last' measure, depicted as line b.

The norms for sustainability are based on the 'negligible risk levels' for concentrations (in air and in water) and the corresponding 'fate analyses' (the link between concentration and emissions).

For further details on these prevention costs, see Chapter 2.



Fiigure 7.3. The way the marginal prevention costs are calculated from emission prevention measures for a certain region.

7.3.3 Implications of the EVR on product portfolio strategy

Progressively, industry is facing the slow but inevitable internalization of environmental costs which are currently external to the costs of production. The rate of this process is unpredictable, but the transformation process as such seems to be inevitable. The eco-costs of a product are a norm for the magnitude of the impact this trend of internalization might have on future product costs.
The eco-costs/value ratio is therefore a measure for the sustainability (eco-efficiency) of a product (see Chapter 5).

With regard to product portfolio management of companies, the EVR model shows the clear implications in the matrix for product-service systems of Figure 7.4.

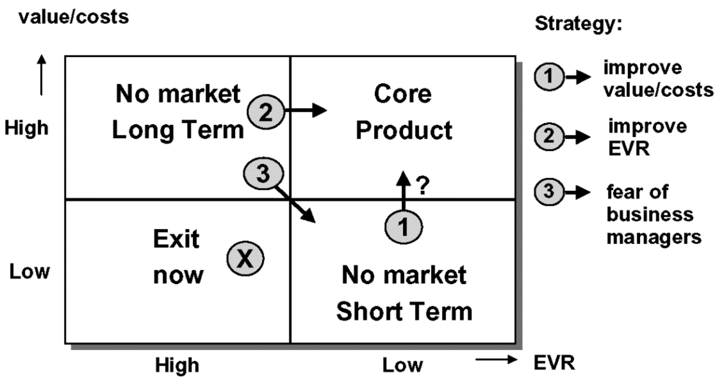


Figure 7.4. Product portfolio matrix for product strategy of companies.

The basic idea of this product portfolio matrix is the fact that each product-service system is characterized by:

- Its short term market potential: the value/costs ratio.
- Its long term market requirement: low eco-costs.

In terms of product strategy, the matrix results in 3 strategic directions:

1. Enhance the value/costs ratio of a sustainable design with low eco-costs to make it fit for short term introduction in the market.
2. Lower the eco-costs of current successful products to make it fit for future markets.
3. Abandon products that combine a low value/costs ratio with high eco-costs.

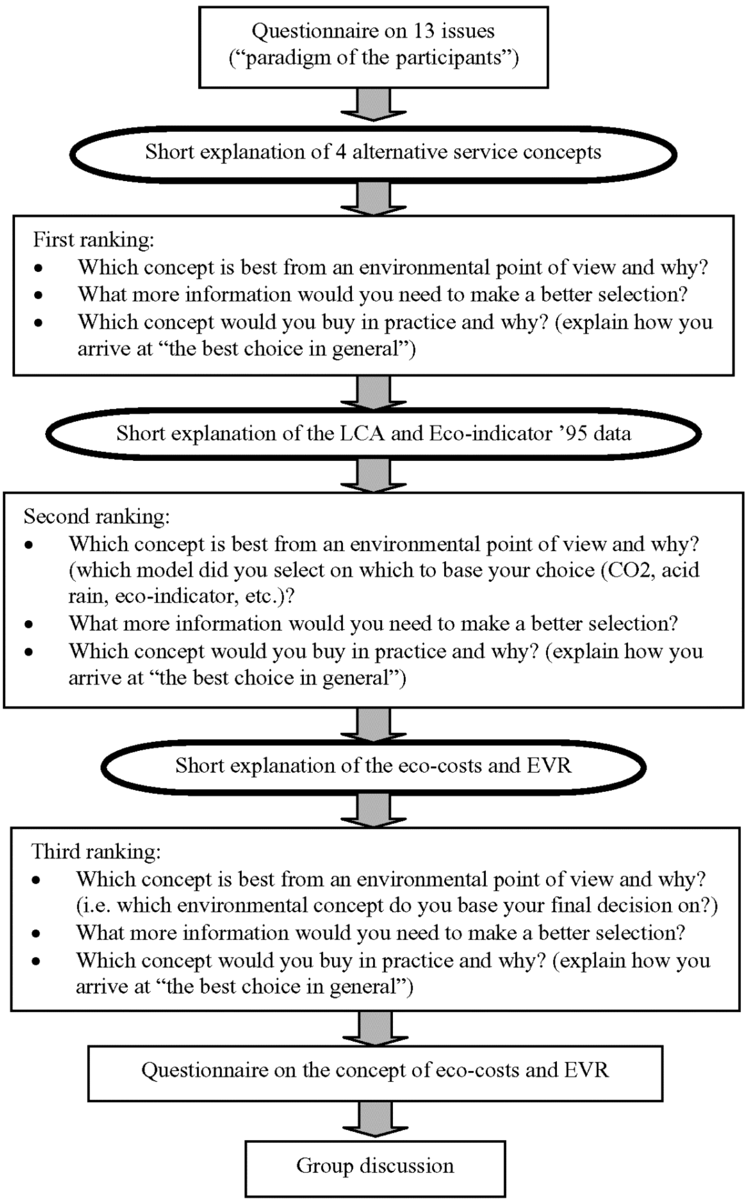
7.4 The experiment

7.4.1 The design of the experiment

The basic idea of the experiment was to provide the participants stepwise with more information on the environmental aspects of the 4 after sales concepts, and monitor whether the opinion of the participants would change as a reaction to this information and how.

The programme of one session had a duration of 4 hours and comprised the steps shown in Figure 7.5.

Figure 7.5. The flow-chart of the experiment.



The experiment was lead by an independent facilitator, and was held by means of the Group Decision Room Computer System of the University of Delft. The room is like a normal meeting room, however, each participant has his or her own computer terminal to type in the answers to the questions (for more information on this Group Decision Room Computer System and the way experiments are designed for this system (Kruijsen, 1999).

The advantage of such a computerized decision system is that the voting, ranking and comments are done anonymously, so without interference (influence) of the other participants. The comments were labelled in the computer with code names, in order to be able to track the individual comments and decisions throughout the session.

The disadvantage of such an approach of anonymous participants however is that specific characteristics of the individual participants (like age, education, etc.) were not known. It was only after the session that we realized (by studying the comments) that experts reacted differently in comparison with non-experts, causing major differences between the governmental group (100% experts) and the other groups (approx. 20% experts).

Although the real information on the environmental aspects is very complex by nature, the concepts were shown in an extremely short time span. Only 5 minutes to explain the basic concept of an LCA and the eco-indicator ’95, 5 minutes to explain the concept of eco-costs (no explanation of how these costs are calculated) and less than 5 minutes for the EVR (eco-costs versus value charts). So especially on the EVR concepts, hardly any time was spent to reflect on it. Many aspects were ‘thrown on the table’ just to check what was ‘understood intuitively’. Another motive to keep the explanation very short was to avoid a situation where participants would have got the feeling that the EVR was ‘promoted’.

The 3 groups received the same information (so the information was not ‘adapted’ to the group). Only the final discussion was focussed on the primary interest of the group, and was therefore different for each session.

With regard to the first, second and third ranking of preferences (Figure 7.5), the questions to be tested were:

- Do people change their preferences when they are confronted with data on sustainability? (Evaluated from the question, “Which concept do you prefer in general?”).
- Do people accept the outcome of a certain model of environmental calculations, after a very short description of the model? (Evaluated from the question, “Which concept of after sales service - is best from the environmental point of view?”)
- Do people change their minds when they are confronted with the concept of eco-costs and EVR after being confronted with the concept of the eco-indicator ’95? (They might become confused when they discover that there are more models to assess the sustainability of a product; do they switch their opinion within such a short time span? Do they prefer the eco-costs and do they accept it?)

- Do people feel that they need more information to choose (at each step of the programme), and if so what kind of information? (evaluated from the question, "What more information do you need to make a better selection?)

7.4.2 Four concepts of after sales service and maintenance of an induction-plate cooker

To be able to conduct the experiment, four alternative product-service combinations were designed.
The product chosen for this experiment is an induction-plate cooker, a 'high quality' product with a premium price (approx. 1800 €) which can be purchased with a 'full guarantee' for 10 years. The service which is chosen for the experiment is the after sales service with this product.

For the experiment we developed 4 different types of hypothetical service concepts, described below.

A. 'Conventional', being the classic type of after sales service (repair)

- In case of a break down, the client calls the company.
- The telephone operator will ask what the problem is.
- The after sales service planning department will schedule the local service engineer within 24 hours
- The logistic system will deliver the required parts overnight in the van of the service engineer
- The engineer is able to repair the induction plate cooker in 70% of the cases;
- In 30% of the cases he needs to visit a second time because he has not been able to repair the product the first time.

B. 'The first time right', being a situation where 100% of the cases are repaired the first time

- By adding the right diagnostic software to the product, the telephone operator knows exactly what is wrong.
- The planning department knows the repair time.
- The logistic system will bring the right parts.

Note. An induction plate cooker has already a lot of control software, so adding diagnostic software can be done relatively cheaply (4'5 to 9 € extra per cooker, which is less than 0.5% of the price).
Major advantage: the client will not be disappointed and there will be less pollution since the service engineer will drive less kilometres.

C. 'Easy to repair by the client'

- The product will be of a modular design (easy clips, screws for click-on and plug- in),

- A repair guide on the web site and a help desk will guide the customer through repair actions,
- Ordering new parts (= modules) by e-mail or at the help desk, delivery by post next morning.

Note. Making the product modular is estimated to add 35 € per cooker to the sales price.

Major advantages: the client doesn't need to stay home for the service engineer, and there will be less pollution since there is no need for service engineer kilometres.

D. 'Designed for less maintenance', reducing maintenance by 60%.

- It appears that 60% of all repairs is caused by the failure of 2 specific circuit boards,
- It is possible to design these parts trouble free either by adding back-up boards, or by heavy testing,
- However the price for this solution is about 180 € (10% of the purchase price) extra.

Note. "Maintenance free" is not possible without a price increase of more than 50%.

Major advantage: reliable product; enhanced durability of the product.

7.4.3 The data on the four concepts

Data were derived from the existing situation of the after sales service. Based on this data, estimates were made on the alternative solutions such as the required additional personnel and investments for each department: the call centre, the logistic departments, the service engineers and the administration and "overheads". Furthermore operational data were gathered such as number of repairs per day per engineer, average kilometres per client, characteristics of the repairs, average costs of parts, etc. (see Table 7.2).

The LCA data of Figure 7.6 and 7.7 were calculated by means of the Simapro computer program (www.pre.nl).

	Chance of repair in 10 years	Costs of service (a)	Costs of parts (b)	Total costs of Repair (a) + (b)	Extra costs of induction cooker
Conventional	60%	65 €	75 €	140 €	
The first time right	60%	50 €	75 €	125 €	4.5- 9 €
Easy to repair by the client	60%	-	80 €	80 €	35 €
Designed for less maintenance	24%	50 €	75 €	125 €	180 €

Table 7.2. Indicative data on the costs of repair of an induction plate cooker. Price levels 1999.

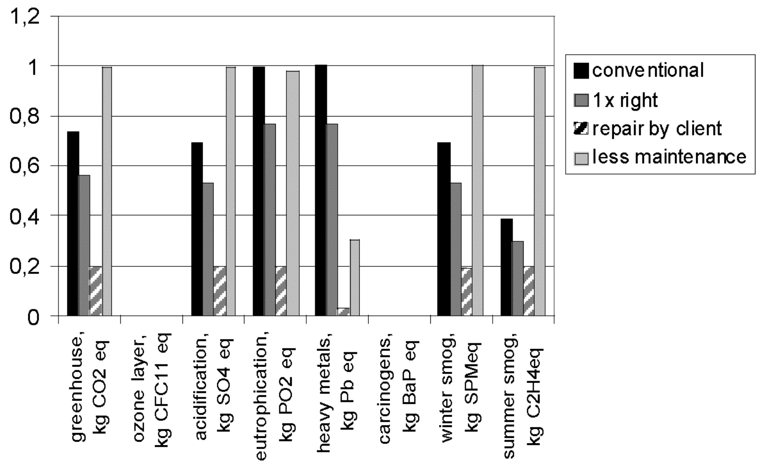


Figure 7.6. The relative emissions of 8 pollution classes for the 4 maintenance concepts.

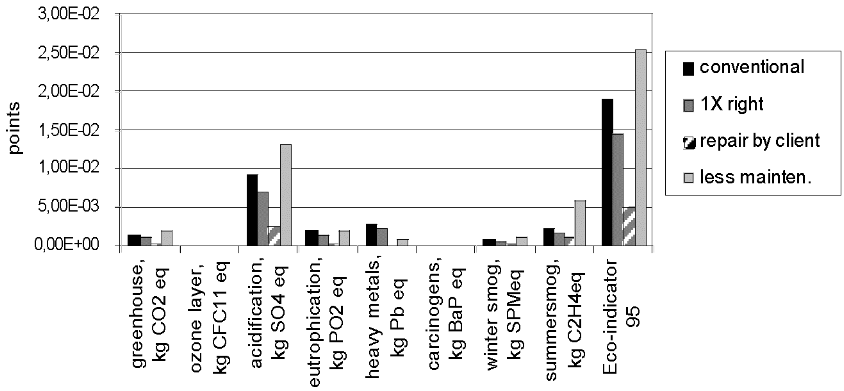


Figure 7.7. The emissions in 'points' of the eco-indicator '95 for the 4 maintenance concepts

For the third part of the experiment, EVR data were calculated for the four alternatives and depicted in EVR charts.

Figure 7.8 depicts the eco-costs and the costs of the various activities which are involved in the repair of the induction plate cooker in the conventional way (as described in the previous section):

- The preparation, including the call centre, the planning and the logistics of the parts.
- Driving to and from the client of the service engineer.
- Repair of the cooker at the home of the client.
- The overheads of the organization.

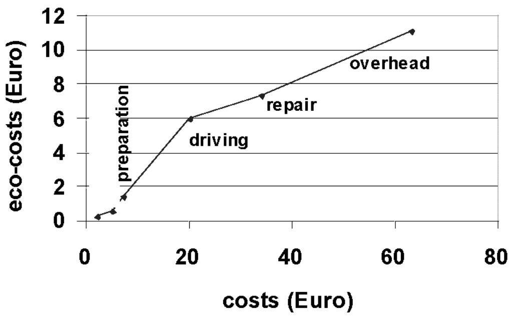


Figure 7.8.. The eco-costs versus costs chart of conventional repair of the induction plate cooker. Price levels 1999.

In Figure 7.9 the eco-cost charts are shown for the four alternative concepts of after sales:

- For "first time right" the savings are in driving, repair and overheads.
- For "repair by client" there is no driving and repair by the service engineer, but the cooker is more expensive and contains more material (the 'last leg' of the line).
- For "less maintenance" there are 60% savings on the repair (first time right) but price of the cooker is 3% – 10 % more expensive and contains more material (the 'last leg' of the line).

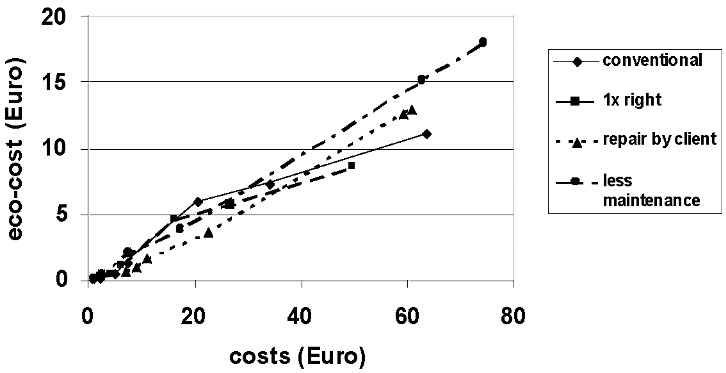


Figure 7.9. The eco-costs versus costs chart of the four concepts of after sales service of the cooker. price levels 1999

7.5 The results of the experiments

7.5.1 Ranking

The results of the first, second and third ranking of preferences for the session with the **consumers** and the session with **business representatives** is depicted in Figures 7.10a, 7.10b, 7.11a and 7.11b.

The consumers group and the business representatives group were quite similar in their choices on ranking of "best choice for the environment" (top score is 4; least score is 1), see Figures 7.11a and 7.12a. They both changed their opinion in each ranking session. They both started with the "guts feel" that "less maintenance" was better for the environment. In later ranking sessions they realized that there was a heavy penalty for it in the extra material required in the cooker.

Detailed analyses of the third ranking session of the consumers group, showed that only 1 out of the 9 participants had chosen for the eco-indicator model instead of the eco-costs model on which to base their ranking (in the third ranking step it was explicitly asked to decide which environmental model would be used for the final choices). In the business group this was only 1 out of 8 participants.

Figure 7.10a. Consumers group environmental ranking.

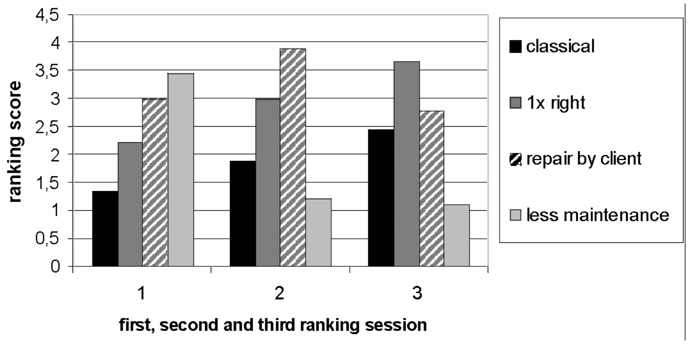
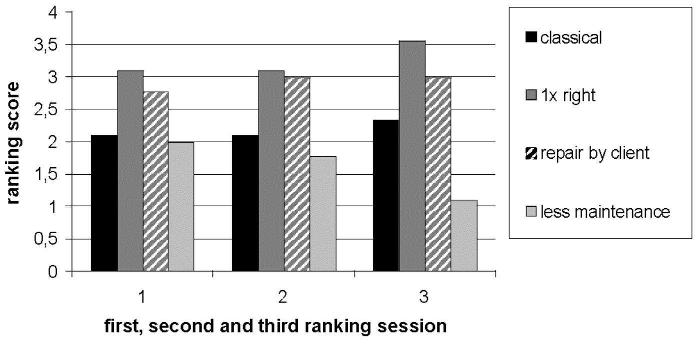


Figure 7.10b. Consumers group preferences.



The difference of ranking on 'the best choice, general' between the two groups was also minor: the main difference is that the consumers ranked 'repair by client' higher than the business representatives.

In both groups 'the best choice, general' was only slightly influenced by the environmental information.

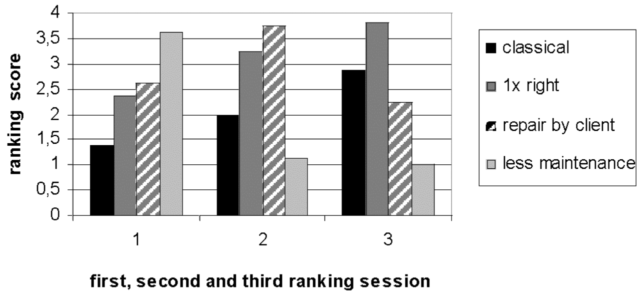


Figure 7.11a. Business representatives group environmental.

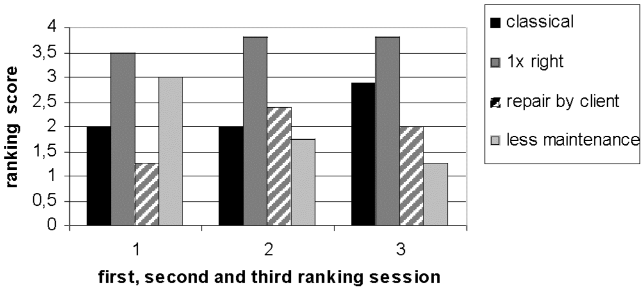


Figure 7.11b. Business representatives preferences.

On the basis of the comments on the question why a certain alternative was chosen, it could be concluded that the environmental aspects play only a secondary role in the choice, as depicted in Figure 7.12. When the value/price ratio already leads to a conclusive choice, customers do not take environmental aspects into consideration anymore. Only when there is no preference on the basis of value/price, do environmental issues help consumers make their final selection.

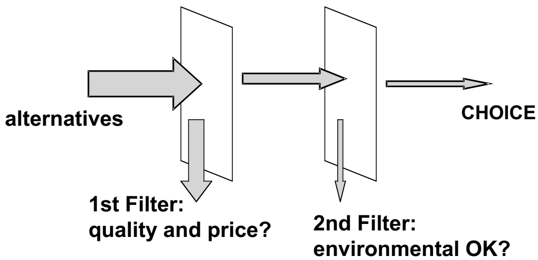
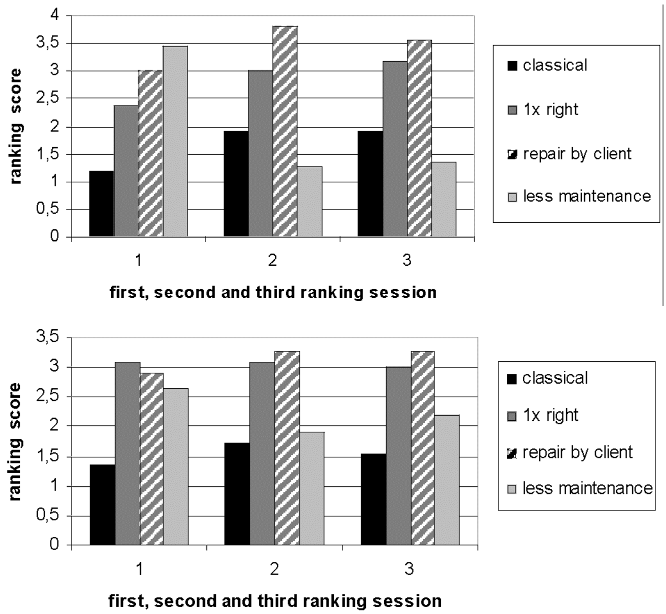


Figure 7.12. Environmental data serve only as a second order filter in the decision of consumers (who are not specialists in the field of environmental issues).

The ranking test of the **governmental group** revealed a totally different pattern than the other two groups. See Figure 7.13.

Figure 7.13
a. Government group environmental ranking.
b. Government group preferences.



The major difference with the other groups is in the third ranking session of the question "what is the best choice for the environment", Figure 7.13a. The third ranking does not differ significantly from the second ranking session. This means that the governmental representatives didn't use the eco-cost model as the preferred model. Analyses of the comments revealed that this seemed to be related with the fact that the participants were all experts in the field of sustainability, and were already acquainted with the LCA theories and with the eco-indicator '95 model. Only 3 out of 11 participants used the EVR chart to make the third ranking.
Furthermore, they tend (as most environmental experts do) to "place sustainability above economy" for their personal purchase decisions, so the "double filter model" of Figure 7.12 did not apply to this group.

Since 3 out of 11 participants used the eco-cost versus cost chart to make the third ranking, the comments of the 8 participants who preferred the eco-indicator 95 data were analysed. People who rejected the new model fell into one of the four following categories:

- I don't accept a monetary calculation since it is not allowed to compare ecology with economy; the choice for ecology is a fundamental one, regardless of the economic consequences to reach sustainability.
- I see a new method which might be interesting, but I don't see yet the consequences of the model, so I reject it for the time being.
- I want to know the details of the model first before I can accept it.
- I am used to the Eco-indicator '95, so I don't see why I should accept a new model.

7.6 Conclusions

In order to reach a sustainable society it is important that government, business and consumers understand the concept of eco-efficiency. For this they need information on which to base their decisions. Current environmental information, like LCA, fails to provide the answers in the right form to stakeholders in terms of decision support. The new eco-costs/value model aims to solve this problem but still needs to be communicated to the stakeholders to be understood and accepted by them.

The experiment revealed that the consumers and business representatives (non-experts) accepted the new model, even after a short explanation. They accepted it intuitively on the general philosophy, without a real understanding of the complete model. They understood the idea of eco-costs and the general meaning of the EVR (Eco-costs Value Ratio).
The government representatives (experts), on the other hand, did not accept the new model and stuck with the Eco-indicator 95 information, which was given earlier during the experiment. They did not see the need for a new model (they were specialists after all, not having trouble with LCA data) or did not accept the monetary nature of the model or had many questions before they could accept it. According to the theory of diffusion of innovation (Rogers, 1962), it is common that expert groups stick to existing theories, rather than accepting new ideas. Rogers' studies revealed that these groups can be convinced only by 'opinion leaders' in their own profession.

The general impression of the whole experiment is that:

- Consumers and business managers seem to be helped in their decisions concerning the environment by a single indicator for LCAs; a single indicator in terms of money (costs) has more appeal to them than a single indicator in 'points'.
- The aspect of sustainability plays hardly any role in the decision when a consumer has a strong preference (based on other aspects) for a certain product type.
- However the aspect of sustainability can play a quite important role in the decision when there is no preference on other grounds.

This suggests that a real breakthrough (in terms of impact on sustainability) in green marketing can be expected only when the aspect of sustainability is dealt with in terms of the 'second order filter' of Figure 7.13. Sustainability can be made the distinguishing factor of choice, especially for commodity products and services (note that maintenance is a 'commodity service'). A precondition is that sustainability must be communicated in terms of a reliable indicator (where possible together with a certification system), preferably in terms of money.

It is recommended to test the above conclusions on a bigger, randomly selected, group of people.

8 Road transport of consumer goods[53]

Case: An LCA based calculation on transport of fresh fruit and vegetables from a Dutch greenhouse to a German retail shop

8.1 Abstract

In this chapter, the EVR model is applied on one complex service system, being a transport case.
In most of the Life Cycle Analyses of consumer goods, only the direct emissions of road transport are taken into account, applying direct emission data for diesel trucks per tonkilometer. For consumer goods this is far from adequate since, for most of the cases:

- Distribution of consumer goods is not determined by weight, but by volume.
- Distribution of consumer goods is done by quite complex logistic chains (so-called hub-and-spoke systems) to minimize transport costs.
- Often there is a quite complex interaction between product, transport packaging and logistic distribution system.
- The indirect emissions (manufacturing and maintenance of trucks, trailers and forklift trucks, as well as construction and maintenance of warehouses and roads) are considerable in comparison with the direct emissions (diesel fuel for trucks).

So the LCA of the transport of consumer goods is quite complex, both in terms of logistics and in terms of the complexity related to the so-called *allocation* of systems (trucks, warehouses, etc.) which are partly used by the goods which are transported.

The complexity of the LCA structure requires the use of a 'single indicator'. In this chapter the eco-costs 2007 system of Chapter 3 is used as single indicator.
A computer program has been developed to analyse such transport systems, applying the economic allocation principle to build a 'totally integrated LCA'. It is shown how

[53] The original title: De ecokosten van transport (Brantjes, 1999), or, The eco-costs, the costs and the EVR of road transport of consumer goods (Vogtländer, 2001,A).

the EVR model can be used to simplify calculations, making calculations feasible without the computer model.
Since there is an ongoing debate on the environmental aspects of re-usable crates versus paper board boxes in the Netherlands since 1990, a calculation is given on the transport of fresh fruit and vegetables from a Dutch greenhouse to a German retail shop. Two packaging systems are compared:

- Solid board boxes (made from recycled paper).
- Re-usable plastic crates (made of High Density Polyethylene, HDPE).

From an environmental perspective the analyses shows that:

- The solid board system seems to be better from the environmental point of view.
- Plastic crates for fresh fruit and vegetables should be designed for maximum relative volume content (instead of minimum materials for the crate) to optimize the use of the transport system.

An attempt should be made to design the system of solid board boxes for two or three round trips per box for 'short distance & high volume' applications.

8.2 Introduction

The road transport of products is increasingly becoming an environmental burden, since the volume of road transport is growing at an even faster rate than our economy.
Based on macro-economic statistical data, a recent study (Bos, 1998) on the indirect energy requirements and emissions from freight transport has shown that the 'indirect' emissions are important in comparison with the 'direct' emissions. The 'direct' emissions result from the use of energy (diesel fuel), the 'indirect' emissions are resulting from:

- Manufacturing and maintenance of trucks.
- Construction and maintenance of warehouse buildings.
- Construction and maintenance of road infrastructure.

In the Life Cycle Analysis (LCA) of a product, transport is obviously an element.
However, the emissions of road transport of consumer products cannot be derived from the aforementioned study, since each specific type of consumer product has its own specific type of logistic transport and distribution system where storage and utilization rates of trucks play an important role. Furthermore, the transport costs and emissions for consumer products are in nearly all cases based on volume and not on weight, whereas macro-economic data are based on weight.
So it was decided to make an LCA based analyses of the transport function, where the transport packaging system is an integral part of the logistic distribution system.

Is a solid or corrugated board box better than a plastic crate, since the boxes are made of recycled material (recycled paper) or is a plastic crate better since it is re-usable (durable)? How about the transport and handling of the empty crates? How about the fact that a crate has a poor net volume versus gross volume? How about the pollution of mills for board and recycled paper?
It is generally accepted that retail companies have lower distribution costs when they apply crate systems, but it is also known that crate systems tend to be expensive at the front end of the chain (filling, storage and transport). Does the environmental burden go hand in hand with the costs?
Within what distance is the crate more attractive from the environmental perspective? And what are the key elements to improve the design of both packaging systems?

Since the transport of fresh fruit and vegetables from the Dutch greenhouses is done in re-usable plastic crates as well as in solid (and corrugated) board boxes, this is used as a case in the study.
Germany was selected as consumers market, giving an interesting range of transport distances: Duisburg (Rhurgebied), 200 km; Frankfurt, 500 km; München, 800 km.

Because of the complexity of the logistic system, there is a need to express the results of the several classes of the underlying LCAs in one parameter: a so-called 'single indicator'. We will apply here the eco-costs of Chapter 3. This choice also enables the use of the EVRs as 'allocation' parameters in the LCA model (according to ISO 14044, see Chapter 3).

In the following, first the overall logistic system is described, and then the details of a link in the chain. Thereafter the calculation structure of costs and eco-costs are given, and the results of calculations are discussed in terms of design consequences.
A method is provided to make a quick estimate of the eco-costs of complex logistic systems.

8.3 The transport chain: a hub-and-spokes system

The transport and distribution chain for Dutch fresh fruit and vegetables is a so-called 'hub-and-spokes' system:

- In the first leg the goods are transported from the greenhouse to the warehouse of the auction or export company in Holland (Hub 1), where all fruit and/or vegetables of that day are stored.
- In the second leg the goods are transported to the distribution centre of the retailer in Germany.
- In the third leg the goods are distributed from the distribution centre (Hub 2) to the retail shops.

This system is depicted in Figure 8.1.

Figure 8.1. Structure of the transport and distribution system.

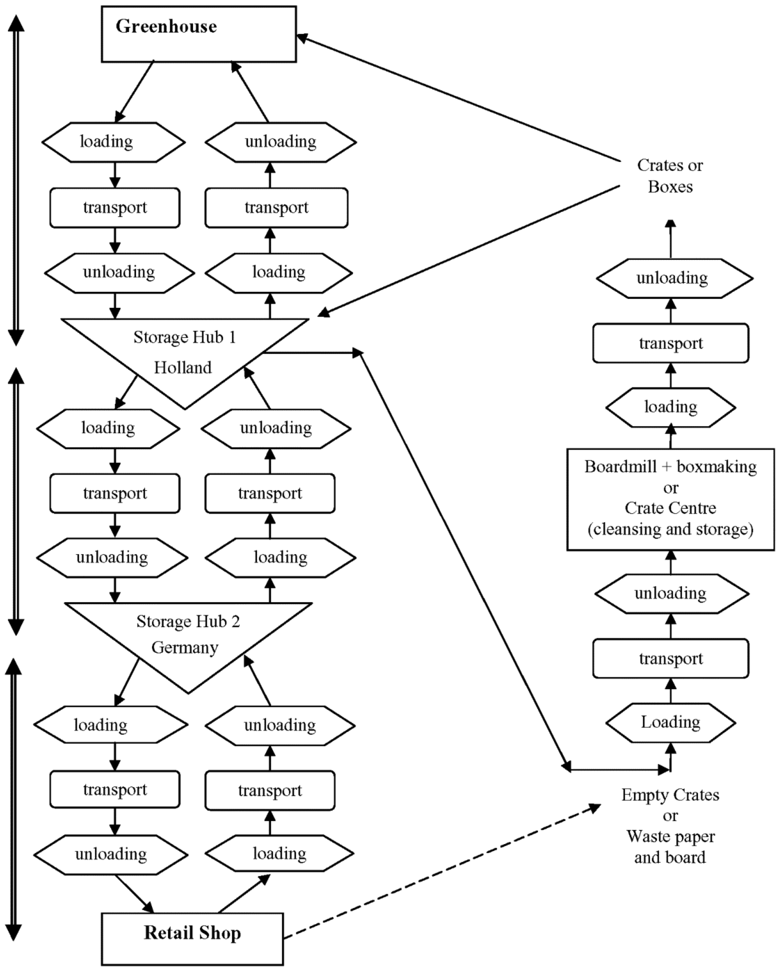


Figure 8.1 also shows the how the crates (and pallets) are returned in the chain. Within the EC, packaging materials have to be returned to the source, so the waste paper (and board) of boxes are returned as well. This might be done via the chain or separately in waste paper chains.

Storage and cleansing of the crates is done at the warehouses of the auctions (extra transport of crates between auctions is only done when the request for empty crates is out of balance with the supply).

Such an hub-and-spokes system is also common for goods other than fresh food. The logistic idea behind it is that in such a system the truckload for the long distance (in this case Holland-Germany) can be maximized. ('hub-and-spokes' refers to a wheel: freight is collected – the 'spokes' – and temporarily stored in a warehouse – the 'hub' –, transported at a high frequency and optimum efficiency to the other hub, stored, and distributed there over the adjacent area – the 'spokes' -).

Most of the international transport companies operate in this way to minimize costs. They run their own warehouses in the hubs: in a well designed logistic system the extra costs of intermediate storage is less than the savings of having a better utilization of the total truck fleet. Since the EVR of transport is higher than the EVR of storage and its related handling, optimization of costs go here hand in hand with minimization of environmental burden (see Section 8.7).

There are many hubs (auctions and export companies) in Holland and many in Germany (every retail company has its own distribution centres). The trucks from Holland to Germany are basically operating as shuttles: the trip back to Holland is either filled with empty crates or, in the case of 'one way' transport packaging, the transport companies try to transport other commercial goods on the trip back to Holland. However, in such a fast and frequent operation it is hardly feasible to arrange 100% payload for the trip back. So the economic feasibility of re-usable crate systems depends on the distance for transport and the availability of other commercial freight for the trip back.

8.4 The structure of one link (leg) in the chain

In order to analyse the logistic system, the structure of one link (leg) in the chain has to be detailed on the level of activities:

1. Pallets with full crates have to be transported from the storage or filling area to the dispatch area by forklift trucks.
2. Pallets have to be loaded by forklift trucks.
3. The truck is driving from place A to place B.
4. Pallets are unloaded by forklift trucks.
5. Pallets with empty crates are loaded with forklift trucks.
6. The truck is driving from place B to place A.
7. Pallets with empty crates are unloaded with forklift trucks.
8. Pallets with empty crates are transported to storage.

This process is depicted in Figure 8.2.

So there are three main activity groups:
Transport.
Loading and unloading.
Storage.

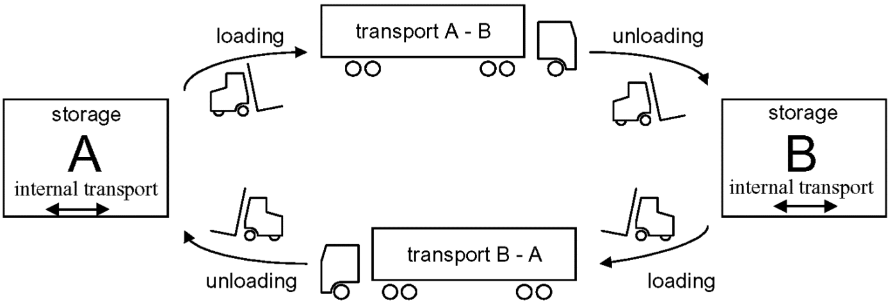


Figure 8.2. The structure of one link (leg) in the chain.

Note. From Section 8.3 it can be concluded that storage A and storage B of the 'transport cycle' of Figure 8.2 might change each cycle. There are many 'storage A' locations in the front end of the chain (there are many greenhouses in Holland) and many 'storage B' locations in the rear end of the chain (many distribution centres in Germany). Since this complexity does not influence the calculations, it is left out in the following analyses.

The main elements of this link of the chain are:

a) The truck.
b) The forklift truck.
c) The warehouse for storage.
d) The road infrastructure.

Each of these elements comprises (the so-called attributes):

- The object.
- The 'direct' energy requirements (i.e. fuel, electricity).
- The related 'direct' labour (e.g. the forklift truck driver).
- 'Indirect costs' such as insurance, interest, etc.

Each object has its own life cycle (value chain):

- The materials required.
- The manufacturing.
- The distribution (of the truck and the forklift truck).
- The use and the maintenance.
- The End-of-Life.

According to this structure, a spreadsheet program has been made to facilitate the calculations. In the input of the spreadsheet program, the activity is defined per main element; the output gives the costs and the eco-costs of the sum of all activities.
In Section 8.5 the general data on the costs and the eco-costs for each main element are summarized. In Section 8.6 it is shown how the costs and the eco-costs of activities in the transport cycle and in the transport chain are calculated from the general data of the main elements.

8.5 General data on the main elements

8.5.1 Truck+trailer, Lorry, and Van

The type of truck+trailer which has been analysed in this study is depicted in Figure 8.3

Figure 8.3. A truck + trailer as normally used in the EC for transport of fresh food.

The costs per kilometre (EVO, 1999, Kuipers, 1998) are calculated in Table 8.1:

- Truck+trailer (net 24 tons, 26 pallets)	0.709 €/km
- Lorry (net 5 tons, 10 pallets)	0.432 €/km
- Van (net 5 m2, 2 pallets)	0.255 €/km[54]

The costs of the driver are about 21 €/hour (Kuipers, 1998). This is based on approx. 2000 driving hours per annum (by Dutch law there is a maximum of 110 driving hours per 2 weeks).

The eco-costs of trucks, lorries and vans are calculated according to the scheme of Figure 8.4 where:

Total eco-costs for each step in the chain = (pollution prevention costs of emissions) + (eco-costs of materials depletion) + (eco-costs of use of energy) + (eco-costs of labour)

The results of the calculations are given in Table 8.2.

[54] It has to be mentioned here that the calculations have been done on the basis of a diesel fuel price of 0.90 € per litre excl. VAT, being the price level of mid 2007 as well as mid 2009 in the Netherlands. The price level fluctuated heavily in the recent years. This new price is approx 60% higher than the original calculation.

Table 8.1. General financial data on trucks, lorries and vans (EVO, 1999, Kuipers, 1998), prices 2007.

All prices excl. VAT; diesel 0.90 €/litre	Truck+trailer	Lorry	Van
	net 24 tons	net 5 tons	net 5 m2
(1) Purchase price (€)	160,000	42,000	15,000
(2) Total distance in life time (km)	1,000,000	350,000	200,000
(3) Diesel fuel (litres/km)	0,33	0,21	0,13
(4) Max. distance one set tyres (km)	100,000	90,000	50000
Eurovignet (€/annum)	1250		
Tax (€/annum)	1100	450	450
Insurance (€/annum)	8100	2950	1200
Interest (€/annum)	3200	1200	350
(5) Subtotal yearly costs (€/annum)	13650	4600	2000
(6) Total distance per year (km/annum)	140,000	70,000	50,000
(7) Max. pallets (1.00 x 1.25m) per trip	26	10	2
Costs per distance (€/km):			
Depreciation =(1)/(2) (€/km)	0.16	0.12	0.075
Diesel =(3)*0.90 (€/km)	0.297	0.189	0.117
Lube oil (€/km)	0.004	0.003	0.001
Maintenance (€/km)	0.09	0.03	0.01
Tyres (€/km)	0.06	0.024	0.012
Yearly costs =(5)/(6) (€/km)	0.098	0.066	0.04
Total (€/km)	*0.709*	*0.432*	*0.255*

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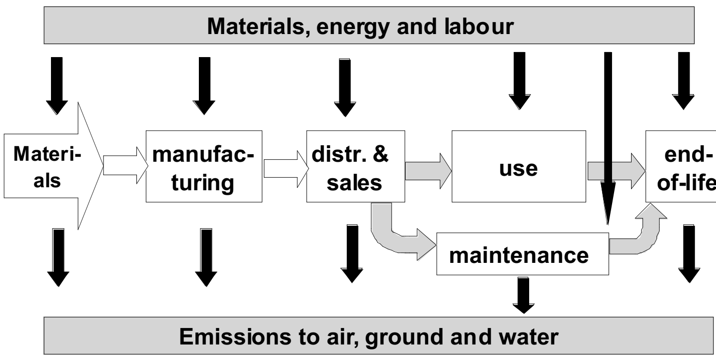

Figure 8.4. The LCA calculation structure.

The eco-costs can be summarized as (see Tables 8.2, 8.3, and 8.4):
- Truck+trailer (net 24 tons, 26 pallets) 0.428 €/km.
- Lorry (net 5 tons, 10 pallets) 0.260 €/km.
- Van (net 5 m^2, 2 pallets) 0.163 €/km.

These costs are excluding the driver and the road (which is the usual approach in LCA).

Table 8.2. Eco-costs 2007 data for truck+trailer.

LCA Truck+trailer, 1.000.000 km eco-costs 2007		eco-costs factor	eco-costs ( )	Total eco-costs ( )
materials truck	(kg)	(/kg)	( )	
steel	6000	0.49	2,940	
PVC	500	0.64	320	
glass	50	0.15	8	
rubber	2000	0.95	1,896	
aluminium (normal trade mix)	200	2.67	534	
copper (normal trade mix)	100	2.44	244	
castwork	1500	0.17	255	
Al extrusion	20	0.12	2	
materials trailer	(kg)	(/kg)	( )	
steel	4800	0.49	2,352	
Aluminium	653	2.67	1,744	
wood	306	0.05	15	
rubber	2300	0.95	2,180	
total materials	*16909*			*12,491*
production&assembly and distribution		EVR	eco-costs ( )	
prod.&assembly ( )	95000	0.23	21,850	
distribution ( )	55000	0.15	8,250	
subtotal truck+trailer				*30,100*
use and maintenance (1.000.000 km)		ecocosts	eco-costs ( )	
diesel (litres)	330000	1.05	346,500	
tyres (kg)	20000	0.95	19,000	
maintenance ( )	90000	0.2	18,000	
total use phase				*383,500*
End of Life (without recycling)	(kg)	ecocosts	eco-costs ( )	
landfill (kg)	16909	0.118	1,995	
total EoL worst case scenario				*1,995*
subtotal excluding diesel				*81,586*
Total eco-costs truck+trailer, 1.000,000 km, incl diesel				*428,086*
Driver at 70 km/hr	( )	EVR	eco-costs ( )	
1000000/70 km = 14285 hours				
21 /hr -> 14285x 21= 300.000	300000	0.05	15,000	15.000
	(km)	ecocost (/km)		
Road (1.000.000 km, 0,09 per km)	1000000	0.09	135,000	135,000
TOTAL truck+trailer (1.000,000) incl. diesel, driver, road				*578,086*

Weight of materials in the second column of Table 8.2 are derived from (Kuhndt, 1999, Bos, 1998) and own calculations. The eco-costs data per kg and per € are from www.ecocostsvalue.com tab data, file 'LCA data on products, services and energy systems' (derived from LCIs of Ecoinvent v2 and Idemat 2008 by Simapro).

Table 8.2 shows the eco-costs of the driver and the road. The cost of the driver is about 21 €/hour (Kuipers, 1998). This is based on approx. 2000 driving hours per annum. The EVR is in this case estimated at 0.05, so the eco-costs of the driver is estimated at 1.05 €/hour.

The calculation of the eco-costs of the road is given in the next section.

Table 8.3. Eco-costs 2007 data for a lorry.

LCA Lorry 350.000 km eco-costs 2007		eco-costs factor	eco-costs ( )	total eco-costs ( )
materials lorry	(kg)	(/kg)	( )	
steel	2500	0.49	1,225	
PVC	200	0.64	128	
glas	30	0.15	5	
rubber (SBR)	500	0.95	474	
aluminium (normal trade mix)	300	2.67	801	
copper (normal trade mix)	50	2.44	122	
wood	100	0.05	5	
machining	1750	0.12	210	
castwork	750	0.17	128	
copperwire	50	0.04	2	
total materials	*3680*			*3,099*
production&assembly and distribution		EVR	eco-costs ( )	
prod.&assembly ( )	24000	0.23	5,520	
distribution ( )	14000	0.15	2,100	
Subtotal lorry				*7,620*
use and maintenance (350.000 km)		ecocosts factor	eco-costs ( )	
diesel (litres), 0,21 l/km	73500	1.05	77,175	
tyres (kg)	500	0.95	475	
maintenance ( )	10500	0.2	2,100	
total use phase				*79,750*
End of Life (without recycling)	(kg)	ecocosts (/kg)	eco-costs ( )	
landfill (kg)	3680	0.118	434	
total EoL worst case scenario				*434*
subtotal excluding diesel				*13.729*
Total lorry, 350.000 km, incl diesel				***90.904***

Table 8.4. Eco-costs 2007 data of a van.

Van, 200.000 km				total
eco-costs 2007		eco-costs	eco-costs	eco-costs
		factor	( )	( )
materials Van	(kg)	(/kg)	( )	
Steel	1550	0.49	760	
PVC	125	0.64	80	
Glas	25	0.15	4	
SBR	200	0.95	190	
Aluminium	70	2.67	187	
Copper	30	2.44	73	
Machining	1050	0.12	126	
Castwork	500	0.17	85	
Copperwire	30	0.04	1	
Aluminium extrusion	70	0.12	8	
total materials	*2000*			*1,514*
production&assembly and distribution		EVR	eco-costs ( )	
prod.&assembly ( )	9000	0.23	2,070	
distribution ( )	5000	0.15	750	
Subtotal Van				*2,820*
use and maintenance (200.000 km)		ecocosts factor	eco-costs ( )	
diesel (litres), 0,13 l/km	26000	1.05	27,300	
tyres (kg)	200	0.95	190	
maintenance ( )	3000	0.2	600	
total use phase				*28,090*
End of Life (without recycling)	(kg)	ecocosts (/kg)	eco-costs ( )	
landfill (kg)	2000	0.118	236	
total EoL worst case scenario				*236*
subtotal excluding diesel				*5,360*
Total Van, 200.000 km, incl diesel				*32,660*

8.5.2 Road infrastructure

The 'embodied energy' of road infrastructure in the Netherlands has been studied at IVEM of the University of Groningen (Bos, 1998). This study has been based on the following macro-economic data for the year 1990 in the Netherlands:

- Total vehicle 'distance × load' by trucks: $47.12 \cdot 10^9$ tonkm/year
- Load factor of utilization of the total truck 0.5

fleet for 'long distance'

- Total embodied energy in road infra. (incl. maintenance) [55] 888 PJ
- Depreciation emb .energy over 50 years 17.8 PJ/year

The above data result in an embedded energy in road infrastructure of 0.38MJ/tonkm Since the majority of the embodied energy stems from the energy used by road transport during the construction phase, and since the major part of that is the use of diesel, the embodied energy is directly converted to eco-costs by the price for sustainable energy for diesel: 29.87 €/GJ (see Table 3.2).
So the eco-costs of road infrastructure[56] can be estimated as:
'maximum load' × 'load factor' × 'embedded energy in roads' × 'eco-costs of sustainable energy'
Which results in the following data of eco-costs of road infrastructure for a truck + trailer:
24 (ton) × 0,5 × 0.378 (MJ/tonkm) × 0.02987 (€/MJ) = 0.135 (€/km)

The above allocation methodology can neither be applied to vans nor to lorries. The main reason is that this methodology is based on macro-economic long distance transport data. Vans are for short distance distribution only, where data in tonkm do not apply. It seems to be reasonable to develop an allocation methodology for vans as part of a methodology for passenger cars. This was, however, beyond the scope of this study.
In line with the general purpose and the general philosophy of the calculation method of this study however, the allocation of the 'eco-costs of road infrastructure' to lorries and vans has been based on the maximum number of pallets which can be carried (an arbitrary, but logical choice):

- Lorries 0.05 (€/km)
- Vans 0.01 (€/km)

8.5.3 Forklift truck

General financial data on forklift trucks are provided in Table 8.5. The costs of a forklift truck are 3.01 €/hour (Brantjes, 1999, Caterpillar, 1999).
The eco-costs of forklift trucks are 1.00 €/hour. The calculation is summarized in Table 8.6.

The costs of the driver are about 19 €/hour. The EVR is in this case estimated at 0.05, so the eco-costs of the driver is estimated at 0.95 €/hour.

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[55] The total embedded energy in roads in The Netherlands is estimated at 3471 PJ, of which 888 PJ has been allocated to trucks (Bos, 1998).

[56] Note that the eco-costs of the embodied energy is the major part of the total eco-costs of infrastructure. A quick estimate shows that the total eco-costs might only be 20% higher. A detailed LCA analysis is recommended.

Note that these costs and eco-costs are negligible in the transport chain of our case.

Figure 8.4. Forklift truck.

Table 8.5. General financial data on forklift trucks. Prices 2007.

Forklift Truck Electricity 0.27 €/kWh	( ) price level 2007
(1) Purchase price (€)	25,000
(2) Total life time (years)	15
(3) Total life time (hours)	25,000
(4) Average operating hours per day	10
(5) Occupancy rate	70%
(6) Power cons. During oper. (kWh/hour)	5.1
(7) Battery life (hours)	6,250
(8) Tyre life (hours)	8,300
(9) Maintenance costs per annum (€)	1,050

Costs per hour (€):	
Depreciation =(1)/(3) (€/hour)	1.00
Electrical power =(6)*0,27 (€/hour)	1.38
Maintenance (€/hour)	0.63
Total (€/hour)	*3.01*

Table 8.6. Eco-costs 2007 data of a forklift truck.

Fotklift truck, 25.000 hr eco-costs 2007		eco-costs factor	eco-costs ( )	total eco-costs ( )
materials forklift truck	(kg)	(/kg)	( )	
steel	2250	0.49	1,103	
lead	1200	0.857	1,028	
sulfuric acid	2800	0.137	385	
SBR	90	0.95	85	
copper	50	2.44	122	
plastic (take POM)	40	1.11	44	
castwork	1000	0.17	170	
machining	1250	0.12	150	
copperwire	50	2.44	122	
Total materials	*6430*			*3,209*
production&assembly and distribution		EVR	eco-costs ( )	
prod.&assembly ( )	13000	0.23	2,990	
distribution ( )	7000	0.15	1,050	
subtotal forklift truck				*4,040*
use and maintenance (25.000 hr)		ecocosts factor	eco-costs ( )	
electrical power (5.1 kWh/h)	127500	0.109	13,898	
tyres (kg)	90	0.95	86	
maintenance ( )	15000	0.20	3,000	
total use phase				*16,983*
End of Life (without recycling)	(kg)	ecocosts (/kg)	eco-costs ( )	
landfill (kg)	6430	0.118	759	
total EoL worst case scenario				*759*
Total Forklift truck, electrical				*24,991*
Total Forklift truck, electrical		*eco-costs per hour*		*1,00*

8.5.4 Warehouse

General financial data on a warehouse of 920 pallets (conventional storage in racks, 4 high) are provided in Table 8.7. The costs of storage are:

- Unconditioned 56.60 €/ pallet.year
- Conditioned 142.20 €/ pallet.year

The eco-costs have been calculated in Table 8.8:

- Unconditioned 15.79 €/ pallet.year
- Conditioned 42.36 €/ pallet.year

warehouse electicity 0,27 (€/kWh)	Warehouse (unconditioned) (€) price level 2007	Warehouse (conditioned) (€) price level 2007
(1) Investment on building (€)	437,000	490,000
(2) Total life time (years)	25	25
(3) Nr of storage positions for pallets	920	920
(4) Maintenance (€/year)	8,400	21,400
(5) Energy consumption (kWh/year)	21,000	246,000
(6) Energy costs per year (€/year)	5,670	66,420
(7) Interest (€/year)	14,000	16,000
(8) Insurance (€/year)	4,800	5,400
Costs per pallet per year (€):		
Depreciation =(1)/(2*3) (€/pallet.year)	19.00	21.30
Electricity =(6)/(3) (€/pallet.year)	6.16	72.20
Maintenance (€/pallet.year)	9.13	23.26
Interest and insurance (€/pallet.year)	20.43	23.26
Total (€/pallet.year)	*54.73*	*140.02*

Table 8.7. General financial data on warehouses; 920 pallets, 900 m^2 25 years.

Table 8.8. Eco-costs 2007 data on warehousing 900 m^2, 920 pallets, 25 yaers.

LCA Warehouse, unconditioned eco-costs 2007		eco-costs factor	eco-costs (€)	total eco-costs (€)
materials warehouse	(kg)	(€/kg)	(€)	
concrete, reinforced, 660000kg	660000	0.046	30,360	
Fe360, 51000kg	51000	0.487	24,837	
steel sheet, 22000kg	22000	0.487	10,713	
PS, 40kg	40	1.324	53	
			0	
PS foaming, 40kg	40	0.000	0	
steel transforming, 22000kg	22000	0.060	1,320	
steel transforming, 51000kg	51000	0.060	3,060	
total materials	733040			70,343
production parts and construction	(€)	EVR factor	eco-costs (€)	
	437000	0.23	100,510	100,510
use and maintenance (25 years)	(€)	EVR factor	eco-costs (€)	
maintenance	240000	0.2	48,000	
electricity (530000 kWh)	530000	0.109	57,770	
total use phase				105,770
End of Life (without recycling)	(kg)	ecocosts (€/kg)	eco-costs (€)	
landfill (kg)	733040	0.118	86,499	
total EoL worst case scenario				86,499
Total warehouse, unconditioned (25 years)				*363.121*
warehouse, unconditioned	eco-costs, € per pallet,year			15.79
extra electricity (5600000 kWh) for conditioned storage per year	*5600000*	*0.109*	*610,400*	*610,400*
extra building costs per year (€)	2500	0.25	625	625
total extra for conditioned storage	eco-costs, € per pallet.year			*26.57*

8.6 Activity Based Costing calculation for costs and eco-costs of a total transport cycle

A computer spreadsheet program has been developed to calculate the costs and eco-costs of transport, based on the data of the previous section and based on the following input per transport leg for the transport, the loading and the unloading, and for the storage respectively:

- Type of vehicle (truck+trailer, light lorry, van).
- Number of pallets in the vehicle.
- Number of pallets return.
- Percentage of freight in vehicle (other freight might be transported at the same time).
- Percentage of freight in vehicle return (when there is no other return freight 100%).
- Distance in km.
- Waiting time for vehicle at docks for loading and unloading.
- Loading and unloading time per pallet for forklift truck.
- Time for forklift truck for storage in the warehouse (per pallet).
- Type of storage (conditioned or unconditioned).
- Storage time of pallets.

The above set of input data enables an 'Activity Based Costing' calculation for the costs as well as the eco-costs.

Such a calculation has been made for transport of tomatoes and peppers from the Dutch greenhouses to the retailer shops in Germany (Frankfurt).
The main characteristics are:

- For the first transport leg ("feeding"): truck+trailer, distance 50 km at a speed of 30 km/hour, number of pallets 26 (full truck load), storage 1 day at the green-house.
- For the second transport leg: truck+trailer, distance 500 km at a speed of 70 km/hour, number of pallets 26 (full truck load), storage 1 day at the German distribution centre.
- For the third transport leg: truck+trailer, distance 50 km at a speed of 30 km/hour, average number of pallets 21 (80% truck load)[57], storage 1 day at the distribution centre of the retailer.

The calculations have been made for 3 types of transport packaging:

- Plastic re-usable crates, life of 30 round trips.

[57] For the third leg it is obvious that in reality the truck is loaded with a full range of products and not with peppers or tomatoes alone; for the calculation, however, this does not make any difference.

- Solid board boxes ('trays'), 1 trip.
- Foldable plastic crates, life of 20 round trips (because of more damage than rigid crates).

General data on solid board boxes and plastic re-usable crates summarized in Table 8.9.

Table 8.9. General data on transport packaging; Prices 2004.

	Re-usable plastic crates (30 trips)	Solid board boxes ("trays")
Content per pallet	2279 (litres)	2467 litres
Costs of transport packaging per litre	0.0093 (€/litre)	0.0183 (€/litre)
Eco-costs of transport packaging per litre	0.0015 (€/litre)	0.0029 (€/litre)

The reason that the content (net transport volume) per pallet of re-usable crate is lower than the content of board boxes, is because of the wall thickness of the crates. The difference with solid board boxes is about 8.3%. This means that transport by solid board boxes is 8.3% more efficient by volume.
Table 8.9 shows that the costs as well as the eco-costs of a re-usable crate system are approximately a factor 2 lower than the costs of a system with solid board boxes. For this reason, all big retail companies have switched to crate systems for their short distance operations.
However, the 'functional unit' of which the comparison must take place is not 'litres', but the 'transport of litres over a certain distance'. When the transport function is taken as a basis for comparison, the picture becomes totally different, since the transport efficiency of a corrugated box is much better than the transport efficiency of a crate (partly because of the efficiency in volume, partly because of the transport of the empty crates).
In the next section, results of the calculations of the total transport chain will be presented.

8.7 Costs and eco-costs of transport of fresh tomatoes and peppers from Holland to Germany; results of the calculations

The total transport chain (transport cycle) of the calculation is summarized in Figure 8.5, where the crates are cleaned and stored at the warehouse of the auction. The results of the calculation for plastic crates as well as solid board boxes are depicted in Figure 8.6 under the assumption that there is freight (70% load) available which can be transported in the return leg from Germany to Holland (leg 2).

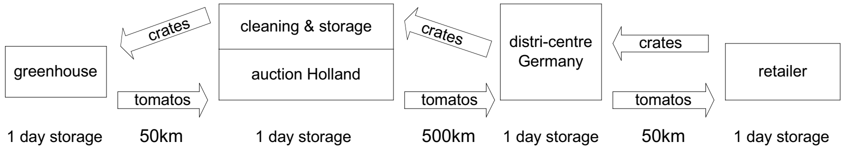

Figure 8.5. Transport cases of fresh tomatoes and peppers from Holland to Germany.

In the case of the plastic crates, the empty crates must be brought back, so there is no payload back. Figure 8.6 depicts also the third line for foldable crates: here 70% payload on the trip back is assumed.

The main conclusions of Figure 8.6 are:

- The solid board system has lower eco-costs than the plastic crate system, because of a better transport efficiency in leg 2 (the leg Germany-Holland) of 500 km. The costs are break-even with plastic crates for 500 km.
- The solid board system, however, is more expensive than the crate system for distances shorter than 500 km. For the long distances, the solid board system becomes less expensive, because of better transport efficiency.
- The foldable crate system seems not a good solution.

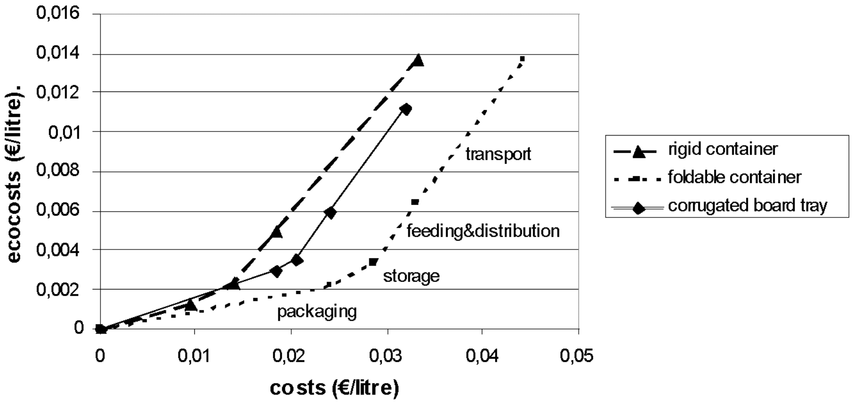

Figure 8.6. Costs and eco-costs per litre net transport volume for the total chain (leg 2 is 500 km). 70% return freight in case of the corrugated box and the folded crate. Prices 2004. Fefco study (Vogtländer 2004,B))

Crate systems have two disadvantages in terms of value for the retailer:

- The perceived quality of the solid board boxes (as trays in the shops).
- The lack of international standardization of plastic crates[58].

With regard to possible improvements of the packaging systems the following observations can be made:

- The crates have apparently been designed to minimize the use of materials: a redesign to maximize net transport volume (using perhaps a bit more material) is

[58] Standardization of crates is a key-element of the system: when there are more types of crates in the system, storage, handling and cleaning of crates become rapidly cumbersome in terms of operations and related costs.

recommended since it will result in less costs and eco-costs of the total chain in terms of transport volume.

- Especially in big retail operations, the solid board boxes are hardly damaged in the chain, so the boxes might be used for two or three trips on average, which will bring down the costs for short distances to a level of the crate systems.

The EVR for each activity in Figure 8.6 has been calculated. These calculations result in the 7 EVR values for transport with truck+trailer (including driver and road):

- Transport packaging, (rigit) crates 0.16
- Transport packaging, solid board boxes 0.16
- Long distance transport (average 70 km/hr) 0.58
- Transport for feeding and distribution truck+trailer (average 30 km/hr) 0.49
- storage systems, unconditioned 0.29
- storage systems, conditioned 0.30
- loading and unloading activities (forlift truck plus driver) 0.10

8.8 Conclusions

For the design of transport systems an integral LCA approach of the total transport chain (cycle) is required to minimize eco-costs. This is because of the high interaction of the system components: the packaging system, the transport system and the storage system. Efficient use of transportvolume plays a key role as well as the re-use of packaging materials.

The eco-costs for the solid board box system appeared to be lower for all cases, especially when the truck can be used for other freight on the return trip, and for longer distances. So there is no reason from the environmental perspective to prefer plastic re-usable crates, which is an embarrassing conclusion in the light of the discussions in the Netherlands that started in the early nineties, when people thought that more durable = more sustainable.[59.]

From an environmental perspective the analyses show that:

- Plastic crates for fresh fruit and vegetables should be designed for maximum net volume content (instead of minimum materials for the crate) to optimize the use of the transport system (since the eco-costs of transport are more important than the eco-costs of the crates).
- For solid board boxes it remains important to lower the emissions during production (the best Dutch manufacturing facilities show already better LCA data than the average data which is used in this chapter).

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[59] In many cases 'durable' (= reusable) does not go hand in hand with 'sustainable', because other aspects of the total system play an important role.

- An attempt should be made to design the system of solid board boxes for two or three round trips per box for 'short distance & high volume' applications, to make them cost competitive for the short distances as well.

9 Recycling of building materials[60]

Four Cases: Concrete aggregate in concrete. Sand extraction at sea. Concrete aggregate in the roads. The mobile crusher.

9.1 Abstract

The environment is an important subject in the construction industry. This chapter deals with the analyses of 4 specific recycling systems, in terms of the following research questions:

1. What is the environmental advantage of replacing gravel in concrete with concrete aggregate?
2. Can the required sand extraction on land be replaced by sand extraction at sea?
3. From an environmental standpoint, can mixed aggregate be better used in concrete than in the roads?
4. What is the environmental advantage of a mobile crusher as opposed to a static crusher?

The analysis of this chapter leads to the following conclusions with regard to the environment:

a) The advantage of using concrete aggregate (rather than gravel) in concrete primarily lies in the reduction of land fill. Differences between emissions are negligible.
b) From an environmental point of view, sand extraction at sea is not more preferable than sand extraction on land.
c) Although it concerns two totally different systems, in the end there is very little difference between using concrete aggregate in concrete and using mixed aggregate in the road.
d) From an environmental point of view, a mobile crusher is preferable to a static one, because the reduction of transport.

[60] The original title: Herbruikbaarheid in duurzaam bouwen, wat is het rendement van vervanging? (Baetens, 2001).

9.2 Introduction

It seems interesting to see whether the EVR model provides a better insight into a number of complex issues concerning the use of raw materials in the construction industry, e.g. replacing gravel in concrete with concrete aggregate, replacing asphalt in the road with mixed aggregate, and replacing sand extraction on land with sand extraction at sea. This also concerns the systems used to produce mixed aggregate from construction waste: can this be better achieved using a mobile installation or a static one?

In analysing the questions concerning the raw materials used, the complete LCA system ('from the cradle to cradle or to cradle') is analysed using the following steps:

1. Extracting materials.
2. Transport.
3. Processing into the final product (concrete or road).
4. Processing at End of Life, i.e. recycling.

The eco-costs for each step comprises of three 'direct' and two 'indirect' components:

- Virtual pollution prevention costs.
- Eco-costs of energy.
- Eco-costs of materials depletion.
- Eco-costs of depreciation.
- Eco-costs of labour.

Two dominating aspects of the eco-costs of land use are taken into account when extracting materials:

- The botanical aspects ('bio-diversity').
- The aspect of 'scenic beauty'.

With regard to the eco-costs of depreciation of equipment, and with regard to trucks, the following data have been used:

1. Dredging equipment (data on eco-costs, see Table 9.1).
2. Grinding equipment (data on eco-costs, see Table 9.1).
3. Concrete production plants (data on eco-costs, see Table 9.1).
4. Asphalt plants (data on eco-costs, see Table 9.1).
5. Shovels for road construction (data on eco-costs, see Table 9.2).
6. Asphalt spreading machine (data on eco-costs, see Table 9.2).
7. Trucks (data on eco-costs database at www.ecocostsvalue.com, 28 tons of payload, however 0.72 litres diesel per km instead of 0.33 litres diesel per km, which results in € 0.059 eco-costs per tonkm).

With regard to the eco-costs of labour, see Section 3.4.4. In this case calculations are based on two types of employees:

- Drivers without an office (eco-costs € 1046 per annum per employee).
- Employees working in a Portakabin, 33 m² per employee (eco-costs € 2080 per annum per employee).

	eco-costs'	lifetime (years)	production (ton)	eco-costs (per ton)
-Gravel dredging equipment take: 20,000 ton steel	$9.8 * 10^6$	15	$1 * 10^6$	0.65
-Grinding equipment take: 20,000 ton steel	$9.8 * 10^6$	15	$1 * 10^6$	0.65
-Concrete production plant take: 30,000 ton steel	$14.7 * 10^6$	15	$0.25 * 10^6$	3.92
-Asphalt plant take: 30,000 ton steel	$14.7 * 10^6$	15	$0.25 *10^6$	3.92

Table 9.1. Indicative primary data on the eco-costs of dredging equipment, grinding systems and concrete production plants.

	Eco-costs	Lifetime (years)	Production (ton)	Eco-costs (per ton)
- Shovel				
take: 1.200 kg steel	€ 588	8	125,000	€ $5.88*10^{-4}$
take: 100 kg PVC	€ 64	8	125,000	€ $0.64*10^{-4}$
take: 20 kg copper	€ 49	8	125,000	€ $0.49*10^{-4}$
Shovel Total	€ 701	8	125,000	€ $7.01*10^{-4}$
- Asphalt spreading machine				
take: 6.000 kg steel	€ 2,940	8	28,000	€ 0.013

Table 9.2. Indicative primary data on the eco-costs of a shovel (2-ton payload for road construction), and a spreading machine, excl fuel and maintenance.

9.3 The advantage of concrete aggregate in concrete

9.3.1 LCA data on gravel, concrete aggregate and concrete

Table 9.3 shows an overview of the eco-costs of gravel and concrete aggregate. This section provides a short summary of the background of these calculations.

Table 9.3. Eco-costs per ton of gravel and concrete aggregate.

eco-costs per ton of:		gravel	concrete aggregate
1. Diesel	Production + combustion	0.38	1.12
2. Extra transport	Barge, 200 km from river Muse	PM	-
3. Equipment	Depreciation,Table 9.1	0.65	0.65
4. Labour	Office heating, energy; Commuting	0.006	0.006
5. Land use	Species richness	0.63	0
	Total	1.67	1.78

It seems that extracting river gravel causes about the same eco-costs as crushing of concrete to aggregate.
Note. The transport of gravel (and concrete aggregate) is a dominating factor as well: gravel, including the transport to the West of the Netherlands (200 km, river-barge), has eco-costs of approximately 3.80 €/ton.

Extracting gravel takes 0.348 litres of diesel per ton. The eco-costs of diesel (1000 ppm Sulfur), including the CO_2 of combustion, and SO_2, No_x and fine dust emissions, is 1.10 €/litre.
As there is plenty of gravel present throughout the world, the eco-costs of depletion can be set at zero.

Crushing concrete requires approx. 3 kWh per ton = 10.8 MJ per ton. With an efficiency of a diesel engine of 30%, 10.8/0.3 = 36 MJ diesel per ton is required. The eco-costs of diesel is 0.031 €/MJ.

Personnel on a gravel extraction installation or at a crushing installation have an available surface area of 33 m^2 per person. The eco-costs per person are therefore estimated at € 2,080.–. On average, there are three people working on each machine.
Annual production of an average gravel extraction installation or crusher amounts to 1 million tons. The eco-costs are therefore € 6.2×10^{-3} per ton.

Gravel extraction naturally leads to areas being converted into lakes (e.g. around the river Maas). With regard to the eco-costs of land use, it is clear that gravel extraction does not destroy these open areas: the EVR model assumes that the lake areas have equal scenic beauty to the landscape that was there before extraction began. The model does not consider any reduction of scenic beauty, so the eco-costs of scenic beauty are set at zero.

The EVR model does include richness of species (biodiversity): where plants once stood, this area is now under water, i.e. a change has taken place in the Species Richness Indicator (SRI), see Section 6.3.1.
In order to calculate the eco-costs of species richness per ton of gravel, we start with the following example:

- Assume that surface A, for example 0.2 km^2 (= 20 ha = 200,000 m^2) is converted from a nature area into a lake area.
- Assume that the original nature area had a value of S_{land} = 235 per km^2 (see Figure 6.1), while the water in the lake area has a value of S = 0.
- Assume that an area of 200,000 m^2 is dredged to a depth of 30 metres. This amounts to 6 million cubic metres of soil. Assuming that 15% of the soil consists of gravel, then this area contains around 900,000 m^3, or approximately 1.44 million tons of gravel.(similar weight of 1,600 kg/m^3).

Using the equation (6.2) now gives:

$$(9.1) \qquad \Delta SRI = 200{,}000 \times 0.94 = 188{,}000 \text{ equivalent } m^2 \text{ nature}$$

The total eco-costs for 'species richness' are € 4.80 per equivalent m^2 of nature (see Section 6.3.1).
Using equation (6.8a):

$$(9.2) \qquad \text{Eco-cost of species richness} = 188{,}000 \text{ equivalent} \times \text{€ } 4.80 = \text{€ } 902{,}400$$

The eco-costs of species richness amounts to € 902,400/1.44 × 10^6 ton gravel = € 0,63/ton.

9.3.2 The reason for replacing gravel with concrete aggregate: less dumping

Table 9.3 shows that, on the whole, there is little difference between gravel extraction and crushing concrete (the difference is not significant, because the accuracy of an LCA is not better than 30%).
However, it would be wrong to conclude that there is no point in reprocessing concrete into concrete aggregate. One should consider the entire life cycle, including the End of Life phase, in which recycling occurs. See Figure 4.5.
With regard to recycling, the EVR model includes two important elements:

- The 'net eco-benefits of recycling'
- The 'added value of recycling'.

The 'net eco-benefits of recycling' are significantly greater than the differences in eco-costs shown in Table 9.3. This is because, if the concrete is not used to produce concrete aggregate (or other recycled material) then it will have to be dumped. Eco-costs for dumping are set in the model at € 118 per ton, see Section 4.5.1. The 'net eco-benefits of recycling' are therefore € 118 per ton more than the difference in eco-costs of gravel and eco-costs of concrete aggregate: € 118 + (€ 1.67 – € 1.78) = € 117.89. See also Section 4.6.
The component value in the eco-costs/value ratio is the added value of the recycling activity. This added value is the saving in dumping costs (approx. € 130 per ton) plus

the market value of the concrete aggregate at the end of the reprocessing phase (approx. € 12 per ton). See also Figure 4.6.

The eco-benefit/value ratio for replacing gravel with concrete aggregate is:
€118/€142 = 0.83.
The eco-costs/value ratio (= –eco-benefits/value ratio) of the replacement is therefore:
–0.83
This is a very beneficial number, as the recycling activities give added value while simultaneously reducing the environmental impact.

9.4 Sand from land and from the bottom of the sea

9.4.1 LCA data on both materials

Table 9.4 provides an overview of the eco-costs of obtaining sand from land and from the seabed. This section gives a short summary of the background to the calculated data.

Table 9.4. Eco-costs per ton of sand from land or seabed.

eco-costs per ton of	:	sand from land	sand from the sea
1. Diesel	Production + combustion	0.38	3.41
2. Extra transport	Specific, but important	P.M	P.M.
3. Equipment	Depreciation,Table 9.1	0.65	0.65
4. Labour	Office heating, energy: Commuting	0.006	0.006
5. Land use	Species richness	0.63	0
	Total	*1.67*	*4.07*

Extracting sand from land takes 0.348 litres of diesel per ton. The eco-costs of diesel including the CO_2 of combustion, and SO_2, No_x and fine dust emissions, is 1.10 €/litre.
Extracting sand from the seabed takes 3.10 litres of diesel per ton.
As sand is available in sufficient quantities, the eco-costs of depletion are set at zero.

Employees on sand extraction installations (both on land and at sea) have a surface area available of 33 m^2 per person. The eco-costs per person are therefore estimated at € 2080.–. On average, there are three people working on each installation.
Annual production of an average sand extraction installation amounts to 1 million tons.
The eco-costs are therefore € 6.2×10^{-3} per ton (= half eurocent per ton).

Extracting sand from land also results in nature areas being transformed into lake areas. This is similar to gravel extraction (see Section 9.3.1): the eco-costs of scenic beauty are therefore set at zero.
However, the EVR model does include reduction of species richness (biodiversity): where plants once stood, there is now just water. The calculation of eco-costs of

species richness per ton of land sand somewhat arbitrarily uses the same calculation as that for gravel: € 0.63 per ton of land sand.

The EVR model sets the eco-costs of land use for sand extraction from the sea at zero, since there is no actual land use. There is detailed data available with regard to the ecological effects and recovery period of both land- and sea-sand extraction. These are listed in the 2nd SOD (Structure Plan for Surface Minerals), which includes the following issues: the size of the ecological effect, the spatial scale of the ecological effect, and the recovery period once the process has ceased.

9.4.2 Land sand or sea sand? A difficult choice

The SOD states a preference for sea extraction. It is almost impossible to see how the large and dynamic bottom of the North Sea could undergo long-term damage from sand extraction. Yet we should ask ourselves whether this preference is not strongly influenced by politics: on land the NIMBY (Not In My Back Yard) attitude of the local population strongly influences political choice.

Based on the total count in Table 9.4 we can conclude that the eco-costs of land sand (€ 1.74 per ton) are lower than those of sea sand (€ 4.96 per ton). However, we should realize that, in absolute terms, both eco-costs are low.

When the value per ton is included in the analysis, the preference for land sand becomes more pronounced: land sand has a market value of approx. € 5 per ton, while sea sand is valued at € 3 per ton.

The eco-costs/value ratio of both materials is:

- land sand: 0.33 (which is in line with the EVR of buildings)
- sea sand: 1.36 (which is very high in comparison with other products).

Based on the EVR model we can only conclude that land sand is the preferred choice.

9.5 Re-using concrete aggregate in the roads

9.5.1 LCA data on road construction

In order to decide whether or not it makes sense to use mixed aggregate for road construction, the complete LCAs of two road construction systems are analysed, based on 1 m^2 of road surface.

a) 200 mm asphalt = 480 kg asphalt (construction without concrete aggregate).
b) 165 mm asphalt = 396 kg asphalt (construction with 250 mm = 475 kg concrete aggregate).

Table 9.5 provides an overview of the eco-costs of the two road constructions. This section gives a short summary of the background of the data.

An employee at an asphalt plant has an area of 33 m² available. The eco-costs per person are therefore estimated at € 2080.–. On average, three people work on each installation. Annual production of an average asphalt plant amounts to 250.000 tons. The eco-costs are therefore € 0.025 per ton.

We assume that asphalt consists of 95% gravel and 5% bitumen. The eco-costs of bitumen is 0.81 €/kg, the eco-costs of aggregate are 0.0038 €/kg. The eco-costs of the truck (28 tons net) is 0.03 €/tonkm (see www.ecocostsvalue.com).

The eco-costs for laying the road are very similar for both types of road construction. See Table 9.2 for a power shovel, and Table 9.1 for a asphalt spreading machine. Fuel consumption for a shovel is 0.075 liter diesel per ton aggregate. Fuel consumption for a asphalt spreading machine is 0.56 liter diesel per ton asphalt. The eco-costs for this type of diesel (1000 ppm Sulfur) is approx. 1.10 €/kg.

Table 9.5. Total eco-costs for the construction of 1 m² road.

	eco-costs of	0 mm mixed aggregate + 200 mm asphalt	250 mm mixed aggregate + 165 mm asphalt
Materials	bitumen	19.44	16.04
	aggregate	1.73	3.23
	subtotal	21.17	19.27
Land use	species richness	0.29	0.54
Transport (30km)	truck 28t	0.86	1.55
	truck driver	0.002	0.004
Road construction	shovel, depreciation	0	3.33E-04
	shovel, deisel	0	0.039
	spreading machine, depr.	0.006	0.005
	spreading machine, diesel	0.30	0.24
Total		*22.62*	*21.65*

9.5.2 Eco-costs/value ratio of mixed aggregate in roads

Table 9.5 shows that, in total, there is very little difference between the two road constructions.
However, it would be wrong to conclude that there is no point in reprocessing concrete into concrete aggregate. One should consider the entire life cycle, including the End of Life phase, in which recycling occurs. See Figure 4.5.
With regard to recycling, the EVR model includes two important elements:

- the 'net eco-benefits of recycling';
- the 'added value of recycling'.

The 'net eco-benefits of recycling' are considerably greater than the difference in eco-costs from Table 9.9: if the demolition waste were not turned into mixed aggregate (or some other recycled material) then the waste would have to be dumped. Eco-costs for dumping are calculated in the model at € 118 per ton. The 'net benefits of recycling' are therefore € 118 per ton more than the difference in eco-costs for roads made with aggregate and roads made with a top layer of asphalt only: € 118 + € 1 = € 119. See also equation (4.6).

The value component from the eco-costs/value ratio is the added value of the recycling activity, i.e. the saving in dumping costs (approx. € 130 per ton), plus the market value of the mixed aggregate at the end of the recycling process (approx. € 12 per ton). See also Figure 4.6.

The eco-benefits/value ratio of replacing asphalt with mixed aggregate is:
€119/€142 = 0.84.
The eco-costs/value ratio (= –eco-benefits/value ratio) of this replacement is therefore:
– 0.84.
This value is almost identical to the value of using concrete aggregate in concrete.

For the eco-costs/value ratio in the model its makes little difference whether the aggregate is used in road construction or in concrete. The most important aspect, as far as the model is concerned, is that dumping has been avoided. In practice, the aggregate will generally be used more for roads due to the extra high quality specifications for using aggregate in concrete.

9.6 The environmental advantage of using a mobile crusher rather than a static one

9.6.1 The issue

Within the framework of reprocessing construction and demolition waste, the question arises as to whether the waste should be crushed into mixed aggregate using a static or mobile crusher. The environmental-technical aspects are naturally taken into account, as well as the economic aspects. When comparing the two types of installations, we need to look not just at the installations themselves, but also at the two processing systems involved. The three most important aspects for the environment are:

a) quality: 10% of the final product (mixed aggregate) from the static crusher does not meet the specifications stated in the Building Materials Decree; the final product from the mobile crusher does meet all requirements
b) transport: transport of construction and demolition waste and mixed aggregate is generally over a distance of around 25 km more, when using a static crusher rather than a mobile one

c) land use: using a static crusher generally results in a considerable stock of construction and demolition waste and mixed aggregate, while this is not case with mobile crushers, which also do not take up any space on a permanent basis (the mobile crusher is located on the building site itself).

The following four sections provide a global analysis of the aforementioned aspects. Points a (quality) and b (transport) are quantified, both in terms of CO_2 emissions as well as eco-costs. Point c (land use) is broadly defined, but only in terms of eco-costs.

9.6.2 Quality

Aggregates from static crushers often do not comply with the Building Materials Decree. It is estimated that 10% is below standard. This can be caused by contamination via the storage and (sometimes accidentally) mixing with materials that are difficult to process. It is extremely rare for aggregate from mobile crushers to fall below legally required standards.
The eco-costs of this quality effect is 10% of € 118 per ton.

9.6.3 Transport

Transport required from the demolition site to the static crusher, as well as that from the crusher to the road construction site, varies for each situation. The following average values are used for the calculations:

- from demolition site to static crusher: 25 km;
- from crusher to road construction site: 25 km;
- total: 50 km.

With mobile crushing installations, most of the construction and demolition waste is processed on the spot. However, there is also a certain amount of waste from neighbouring demolition work. The waste is sometimes reused on the same site, but may also be transported short distances. The following average values are used in the Calculations:

- from demolition site to crusher: 5 km
- from crusher to road construction site: 15 km
- total: 20 km.

There is clearly more transport involved to and from a static installation than a mobile crusher. This also creates extra environmental impact:

extra eco-costs: € 1.8 per ton.

The following data have been used for this calculation:

- the difference in transport distance is 50 – 20 =30 km;
- eco-costs of transport is € 0.059 per ton-kilometre.

9.6.4 Land use

In theory, a mobile installation does not require any land, as against a static installation (with stockpiles). As far as land use is concerned, a static installation is therefore less advantageous than a mobile one.

In the ecocosting system it is, in theory, possible to express land use (i.e. conversion of land) as a number. The eco-costs of land use can be calculated if specific data concerning the location are known. These eco-costs are therefore dependent upon specific planning details, which make it difficult to use a general number for all static crushers in the Netherlands.
In certain conditions it is only possible to give an indication of the eco-costs for land use.

There may be reduced biodiversity.
The mobile crusher is normally positioned, temporarally, on the new building site, so there is no extra reduction on biodiversity.
If a static crusher is located on built areas that were, until recently, used for other industrial activities, there is no reduction in biodiversity as well.
However, if a static crusher is set up on an old industrial estate, there is definite destruction of biodiversity. The biodiversity of old, often neglected, industrial sites, may be considerable (see Table 6.2). In this case, the eco-costs of setting up a static installation can be estimated in the range of 2.30 – 3.60 €/m^2.

The eco-costs of land use per ton of mixed aggregate can now be calculated using the following data (an average for this type of installation):

- production 150,000 tons of mixed aggregate per year;
- economic lifespan of the installation = 10 years;
- surface area of site 150,000 m^2 (500 m^2 × 300 m^2);
- eco-costs for land use approx. 3 €/m^2 (for building on an old industrial site).

Based on the above data, the eco-costs for land use are calculated as € 0.48 per ton of mixed aggregate.

9.6.5 Overall effect on the Netherlands

Table 9.6 shows the difference between the environmental impact of static crushers compared to mobile installations, both per ton of mixed aggregate as well as the total market for mixed aggregate in the Netherlands ($12 \cdot 10^6$ ton per year).

Eco-cost (€)	quality effect	extra transport	land-use	total
per ton mixed aggregate	11.8	1.80	0.3	13.9
total aggregate per annum in the Netherlands ($12 \cdot 10^6$ ton/annum)	1.42E+08	2.16E+07	3.60E+06	1.67E+08

Table 9.10. The extra environmental burden of a static crushing system compared to a mobil unit.

The disadvantage of a static crusher, expressed in eco-costs, is primarily determined by the loss of quality, assuming that 10% of the material produced is below the quality required by the Building Materials Decree and will be rejected and dumped.
A secondary effect is transport, which schould be calculated for each individual case.

9.7 Conclusions and discussion

In addition to the practical conclusions discussed in the previous sections, a number of more model-related conclusions can be drawn with regard to the EVR model:

1. Methods that only consider emissions (e.g. the 'enviromnental profile' and 'environmental measures' that are often used in the Netherlands) are insufficient on which to base a comprehensive decision; the tables show that emissions form just a part of the integral environmental problem.
2. Although the character of the cases is such that it primarily concerns an analysis of the eco-costs, in a number of cases the EVR provides additional insight (e.g. when comparing several different cases).

APPENDICES

Appendix 1 [61]

From: "Our common future", G.H. Brundtland, World Commission on Environment and Development

In the Preface of Brundtland (Brundtland, 1987), page xii:

> "The downward spiral of poverty and environmental degradation is a waste of opportunities and of resources. In particular it is a waste of human resources. These links between poverty, inequality, and environmental degradation formed a major theme in our analysis and recommendations. What is needed now is a new era of economic growth – growth that is forceful and at the same time socially and environmentally sustainable."

In the summary about 'Sustainable Development', page 9:

> "Yet in the end, sustainable development is not a fixed state of harmony, but rather a process of change in which the exploitation of resources, the direction of investments, the orientation of technological development, and institutional change are made consistent with future as well as present needs. We do not pretend that the process is easy or straightforward. Painful choices have to be made. Thus, in final analyses, sustainable development has to rest on political will."

The definition of sustainable development ('sustainability'), as it is used in this book as well:

> "Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (page 43)[62]

Recommendations in Our Common Future:

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[61] The original title: Annex 1a (Vogtländer, 2001, A)

[62] It is now widely recognized by economists that the goal of sustainable development is principally an equity issue. Sustainable development is a requirement to our, and future, generations to manage the resource base such that the average quality of life we ensure ourselves can potentially be shared by all future generations. High levels of eco-efficiency of product-service systems are required to achieve such a 'intergenerational equity'. (Henley, 1997). This book, however, does not touch on the awkward question of the equity within our own generation: the 'intragenerational equity', which is related to the sustainability issues with regard to the poor parts of our world.

"…

- Establish environmental goals, regulations, incentives and standards.
- Make effective use of economic instruments (from 'externalized' towards 'internalized' costs).
- Broaden environmental assessments.
- Encourage action by industry (more action driven than merely regulation driven).
- Increase capacity to deal with industrial hazards.
- Strengthen international efforts to help developing countries.

…..".

See also footnote [63]

Appendix 2 [64]

Background information on norms for sustainability

The relationship between the concentration and the damage

It has to be mentioned here that the relationship between the concentration of a pollutant and its damage is not known for any of the pollutants: is the relation linear? logarithmic? s-curve type? See Figure A.2.1. It is obvious that one point on the curve (the MTR) is much easier to determine than the shape of the total curve.

Most of the models of a single indicator based on the *damage* of emissions, implicitly assume a linear relationship between emission and damage. This requires a linear function through the origin of the concentration-damage curve (see Figure A.2.1) which is not likely to be the situation in reality!

This is one of the basic flaws in damage-based models, when they are applied to a wider range of concentrations (i.e. a wider range of regions), which is normally the case within LCA calculations.

Prevention based models don't apply a concentration-damage relationship: prevention measures have to bring the concentration under the Negigible Risk Level, being 1/100 to 1/10 of the Maximum Allowable Risk Level[65]. Prevention based models apply therefore only *one point* on the damage-concentration relationship of Figure A.2.1.

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[63] In 1990, Daly defined the operational principles for sustainable development:
set harvest levels of renewable resources at less than, or equal to, the population growth rate
set emissions of pollutants at less than, or equal to, the assimilative capacities
make research funding available for substitutes for non-renewable resources, so those substitutes are available on time
minimize the materials and energy requirements of the economy, and identify maximum economic growth
(Daly, 1990, Henley, 1997)

[64] The original title: Annex 2a (Vogtländer, 2001, A)

[65] For Global Warming, the generally exepted norm is a stabilisation of the global temperature at + 2 °C, resulting in approx 50% reduction of CO_2 emissions of 1990, which is approx 60% reduction of CO_2 emissions of 2007.

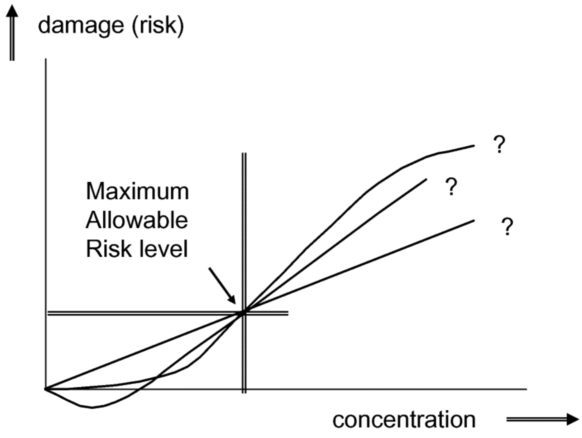


Figure A.2.1. The shape of the concentration-damage curve for pollutants is not known

The relation between emission rates and concentrations
Starting from the situation that the set of 'target concentration levels' is fairly well known, it is, however, not easy to calculate the corresponding 'maximum sustainable emission rate'.

When the rate of decay (or absorption) is known, it is possible to determine the 'maximum sustainable emission levels' because these levels can be calculated for the 'steady state of a closed region' (i.e. the total sustainable emission in that region is set equal to the decay or absorption rate of that pollutant at the maximum allowable air or water concentration). When we assume a (pseudo)first order reaction for decay or absorption, the equation for the 'maximum sustainable emission rate' can be derived:

(A.2.1) first order reaction: $\mathrm{d}c/\mathrm{d}t = \phi/V - k \times c$,

where: c is the concentration of the pollutant (kg/m^3)
t is the time (s)
k is the reaction rate constant for the decay or the absorption(1/s)
ϕ is the emission rate (kg/s)
V is the volume of air or water (m^3)

(A.2.2) for the steady state: $k \times c = \phi/V$
(A.2.3) or, for the maximum sustainable level: $\phi_{max} = k \times c_{max} \times V$

where: ϕ_{max} is the maximum emission rate which is just sustainable (kg/s)
c_{max} is the maximum allowable concentration of the pollutant (kg/m^3)

The MARL for eco-toxicity is a level at which 95% of the potentially resident species are safeguarded when there are no other pollutants. For carcinogens: less than 1 fatal illness per 1 million inhabitants. The Negligible Risk Level is 1/10 to 1/100 of the MARL.

In reality it is more practical to define equation (A.2.3) in terms of the *immission rate*, I, which is defined as the load rate of pollutant (kg/s) per square meter of area, A:

(A.2.4) $I_{max} = k \times c_{max} \times d$

and: $I_{max} = \phi_{max}/A$

where: d represents the average thickness of the polluted layer
e.g. in *water*: the volume of water in a certain area divided by the area surface,
in *air*: for summer smog the height of the inversion layer in the air,
in *soil*: the penetration depth of the pollutant.

De Boer (Dellink, 1997) gives a calculation for the Dutch situation for acidification, eutrophication, summer smog, winter smog and heavy metals (Zn), based on several Dutch calculation models.
Note that damage based models need further calculations to define the relationship between concentrations of pollutants and damage (the 'fate analyses', where regional exposure, individual vulnerability and risk analysis play an important role).

Marginal prevention costs
Most of the experts in LCA study damage based systems, focussing on the problems cause by pollutants. A totally different system to calculate norms for sustainability is based on the 'marginal prevention costs', applied by environmental economists, focussing on the measures which have to be taken.
The basic reasoning behind the marginal prevention costs can be summarized as follows:

- prevention of emission of production processes will require technical measures ('end of pipe' as well as 'process integrated'),
- these measures will cost money (e.g. €/'kg prevented emissions'),
- on the road to a sustainable economy, we will introduce the most cost effective measures first,
- the last and most expensive measure for pollution prevention, required to reach a sustainable economy, has a certain level of costs per 'kg prevented emission': *the marginal prevention costs*.

This is depicted in Figure A.2.2. The measures to be taken form curve a, the marginal prevention costs, are determined by the slope of curve a where the norm for sustainability is met: the slope of line b.
Note that the total required prevention costs to reach the norm are less than the distance to the norm multiplied by the marginal costs (as indicated by line c).
One of the advantages of the marginal prevention costs as a norm is that these costs do not change when measures are implemented, where the 'distance to target' changes continuously in time. See also Appendix 3 for a further explanation.

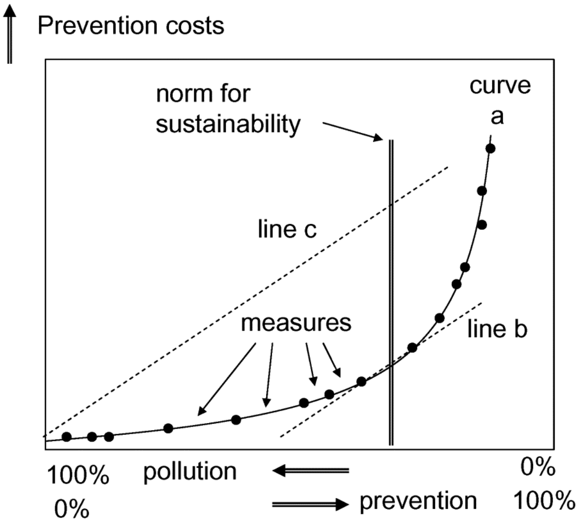


Figure A.2.2. A typical cost curve of reduction of pollution and the marginal prevention costs.

There are several ideas to trigger the required change process on a National, European or even world wide, level. To name a few:

- taxation of emissions ("the pollutant pays") at the level of the marginal prevention costs: companies will apply the cheaper measures, where possible, rather than pay the tax,
- introducing 'tradable emission rights' at the price of the marginal prevention costs: again, companies will use each opportunity to apply measures which are cheaper than these emission rights,
- agree with the industry that they will introduce a list of measures (of which the most expensive measures are at the level of the marginal costs), a so-called 'convenant' to apply the 'best practices',
- force the industry to introduce the measures up to the level of the marginal prevention costs and ban the processes which cannot be accepted (for example the ban on CFCs),
- try to influence the demand side of the market to accept environmentally clean products only.

The prerequisite of these ideas is that our society is prepared to pay the extra costs up to the level of the marginal prevention costs in order to create a sustainable world, and that the measures have to be introduced in the most cost-effective way from the National point of view. See also Appendix 10.

Although the concept of marginal prevention costs is easy to understand, it generates a lot of questions such as:

- what is the level of sustainability for each class?
- what is the 'willingness to pay'[66] for each class?
- will 'economies of scale' and technological innovation make pollution prevention less expensive, resulting in a lower level of marginal prevention costs?
- will economic growth and growth of the population make the sustainability norms harder to reach in future, resulting in a higher level of marginal prevention costs?

The model of the eco-costs (virtual pollution prevention costs) is based on the present situation, and not on the future. The validation of the levels of marginal prevention costs are based on the present state in 'virtual' terms ("what if we had already taken the measures by now"). These calculations are based on the aforementioned 'target concentration levels', which are accurate enough "to begin with".
The last measure for pollution prevention in the calculation of the marginal prevention costs is to be regarded as a 'moving target': the future will bring us the better values of marginal prevention costs (hence the updata from 'eco-costs '99 to 'eco-costs 2007').

The fact that the maximum level of emissions for a sustainable society is not yet fully known, is often used as the main argument to reject prevention oriented models as being too vague. It is used as an argument to choose a damage oriented model for calculations. A rather bizar situation since the main elements for calculations on the basis of damage are not available at all:

- the shape of the concentration-damage curve (where the fact that the curve is non-linear is causing enormous complications in such damage based models),
- a realistic methodology how to compare a fatal illness with dying trees and/or extinguishing species (Finnveden, 1997, Finnveden, 2000, Brengtsson, 2000).

Regionality

A fundamental problem in the calculation model is how to deal with 'regionality' (apart from calculations on the greenhouse effect, since this effect is global).
In areas with a high density of population and industrial activity, stringent and expensive measures are required (to safeguard the sustainable level of the ratio ϕ/V or ϕ/A, see equation (A.2.3) and (A.2.4)). From the mathematical point of view, however, higher emissions are allowed when they are diluted in a higher volume (bigger area). As a consequence, the marginal prevention costs will be higher when the calculation is

[66] The willingness to pay (WTP) must not be confused with the marginal prevention costs. The WTP (and the willingness to accept compensation, WTAC) are based on *valuation of damage*. Although many attempts have been made, there are still a lot of methodological flaws in such a 'non-market valuation', and 'non-use valuation' (Henley, 1997). There have been two major applications of the WTP:
• the valuation of the damage caused by the Exxon Valdes (by the Contingent Valuation Method, CVM)
• the EPS method (Steen, 1996).
One of the main conclusions is that information is crucial for those valuation systems: for many environmental issues, awareness is needed to bring the WPT above the level of the marginal prevention costs (to make the level of marginal prevention costs politically acceptable).

made for the Dutch province of Zuid-Holland, than it is for The Netherlands, or for Europe. Damage based models suffer from the same fundamental problem.
From a philosophical point of view, it seems that dilution of emissions have to be avoided in the real world (so also in the calculation models). At the same time, best practices for pollution prevention should be applied on a global level to prevent 'export' of sustainability problems.

The aforementioned approach means that norms have to be calculated for areas with a high density of population and industry (e.g. The Netherlands, the western part of Germany, the areas of Los Angeles, Tokyo, etc.). The required pollution prevention measures should then be applied world wide, however, not at a cost level which is higher than the marginal prevention costs (example: a windmill is only a good solution in areas with a lot of wind). So the idea of economic feasibility ('cost effectivity') plays an important role in such a model.

Appendix 3 [67]

Why marginal prevention costs instead of total prevention costs

A frequently asked question on the presented theory of the pollution prevention costs (and the eco-costs) is related to the choice of the marginal prevention costs (€/kg) to monetarize the seven classes of emissions. Why not the total costs (€) divided by the total emission reduction (kg)? (Referring to Figures 2.3 through 2.7).

There are three methodological reasons to take the marginal costs as a norm:

- the marginal costs are more stable in time (during the transition towards a sustainable society) than the total costs, so the bases on which calculations are made do not change during such a transition.
- the marginal costs are an estimation of future taxes or tradable emission rights, related to individual products in the event that nothing is done to prevent the related emissions; the marginal prevention costs are therefore relevant for product strategies of designers and business managers.
- the marginal costs are related to specific prevention measures (Best Available Technologies), one for each class of emissions, which makes it plausible that the same marginal prevention costs will apply - in the long run - to all EEC member states.

These three reasons are explained in more detail hereafter.

[67] The original title: Annex 2c (Vogtländer, 2001, A)

Point 1.

The marginal prevention costs are more stable in time (during the transition towards a sustainable society). This is depicted in Figure A.3. for the logical assumption that society will take the most cost-effective measures first.

It is shown in Figure A.3 that the total prevention costs C (€) divided by the total emissions which are still to be tackled, ΔE (kg), will change in time: in the beginning the ratio $C/\Delta E$ is rather low, at the end this ratio grows to the ratio of the marginal prevention costs (the slope of line b).

However, the marginal prevention costs (€/kg), being the slope of line b, will remain constant throughout the total transition process.

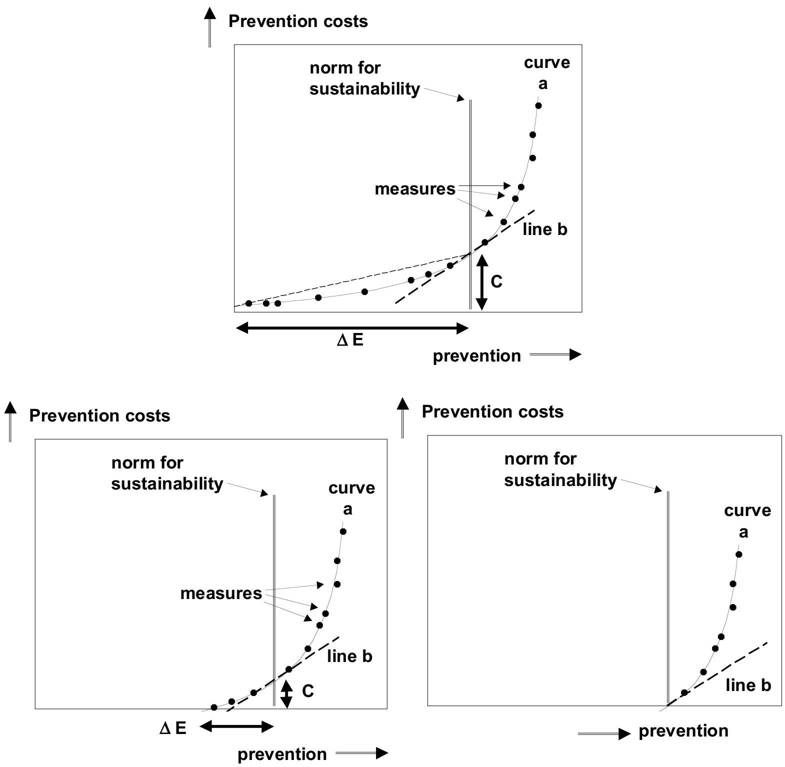


Figure A.3. The transition towards a sustainable society: Upper: current situation, Left: during the transition period, Rright: final sustainable situation).

Point 2.

The marginal prevention costs are an estimation of future taxes or tradable emission rights related to individual products in the event that nothing is done to prevent those emissions. The assumption here is that the government will try to force industry to take the required actions by either taxes or tradable emission rights at the cost of the marginal prevention costs. This is further explained in Appendix 10.

Even when the transition is enforced by other regulations (such as the Dutch 'convenants'), the marginal prevention costs are a good yardstick for business strategies: measures below the prevention costs have to be taken, measures above the yardstick should be avoided since they are not cost-effective. (See Appendix 4 for more details on how, and in what extent, 'external' costs become 'internal' costs of a product.)

Point 3.
The marginal prevention costs are related to a specific prevention measures (Best Available Technologies), one for each class of emissions. This makes it plausible that, in the long run, it can be expected that the same marginal prevention costs (i.e. the same Best Available Technologies) will apply in all EC countries. This is in line with the current policy within the EC with regard to environmental protection (the IPPC Directive): it enforces the member states to take the same measures in order to keep the industrial competitive playing field levelled.
It has to be mentioned here that the measures which are relatively cost-efficient (the measures at the left hand side of curve a in Figure A.3) are in most cases related to industrial activities, since concentrations of emissions in industry are relatively high and therefore easy to tackle (compared with the more diffuse emissions from domestic use). The right hand side of the curve, in combination with the norm for sustainability, is predominantly determined by domestic emissions, and therefore a function of the density of population in a certain region. It is expected therefore that norms within the EC will be governed by norms for densely populated areas (such as the triangle London-Paris-Dortmund).

Appendix 4 [68]

An estimation of future product costs: from 'external' costs to 'internal' costs

An important aspect of our current economy is that the negative value of a product which is related to environmental damage, is not part of the costs of that product. Environmental damage is 'external' to the current cost structure.
Governments will take action to reduce these 'external' costs (by regulations, tax, tradable emission rights, subsidies), which will result in higher product costs.
This annex will explain how, and to what extent, eco-costs become part of future product costs.

The concept of the 'virtual eco-costs' (in short: 'eco-costs') is slightly different from the concept of the 'external costs'. The external costs are related to damage to our environment. The eco-costs are related to the ('marginal') prevention costs, which are required to bring our economy into a state which is sustainable. What both type of costs have in common, is that they are not incorporated in the current costs of products and services (the current 'internal' costs).

[68] The original title: Annex 9a (Vogtländer, 2001, A)

In other words: the eco-costs are related to the costs to prevent damage to the environment. Eco-costs can be regarded as 'hidden obligations'.

In a free market economy, governments can only influence companies to take prevention measures by either imposig tax on emissions, or introducing a system with tradable emission rights, at the level of the marginal prevention costs (see Appendix 10). By doing so, a big proportion of the eco-costs become part of the future product costs. See Figure A.4.

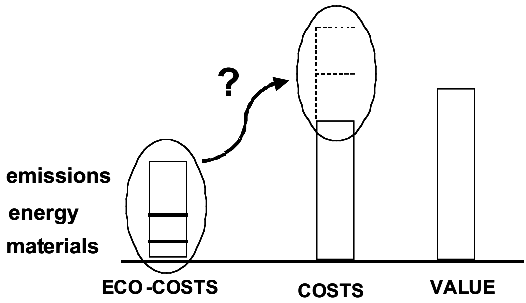


Figure A.4. A proportion of the eco-costs will become part of the costs, when an ecotax system or a system of tradable emission rights is introduced suddenly at the sustainable level.

Note that the total future costs (current costs+ecocosts) can be reduced for most of the current products by introducing cleaner production systems: the current costs will get a bit higher, but the ecocosts will become much lower.

Appendix 5 [69]

The costs-price-value model

In elaborating the concept of eco-efficiency as defined by the WBCSD and the basic idea of the EVR model, it is essential to understand the differences between costs, price and value as they are defined in modern management theories (such as Total Quality Management and/or Continuous Improvement).

The classical management paradigm to describe the function of costs, price and value is depicted in Figure A.5.1.
In the eyes of the producer, profit is a result of the difference between the costs of a product and its price. Managers try to reduce the costs as much as possible and get a price as high as possible.
However, managers know that the end user (consumer) will buy the product only when, in the eyes of these consumers, the perceived value is higher that the price.
In the classical management paradigm, the manager has no choice: when the price gets too high, there will be no buyers, so the only thing he can focus on is reducing costs. In

[69] The original title: Annex 5a (Vogtländer, 2001, A)

this paradigm, measures for environmental protection add costs, so have to be kept to a minimum.

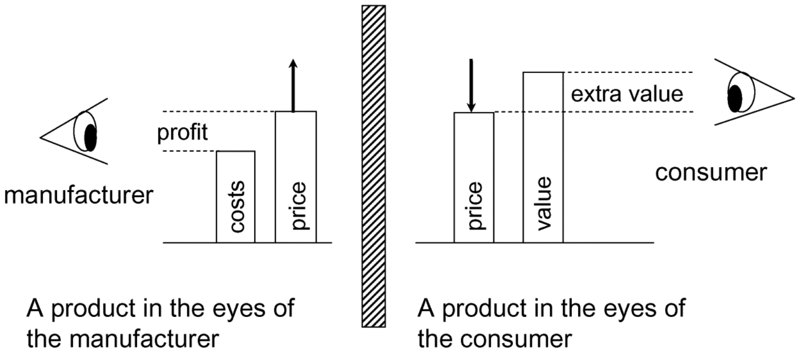


Figure A.5.1. The classical paradigm is "price driven", which leads to "cost cutting".

In the modern management approach, the strategic focus is on the ratio of value and costs, as is depicted in Figure A.5.2. A big difference between value and costs create a variety of strategic options for setting the right price (more profit by optimization of margin per product versus sales volume).

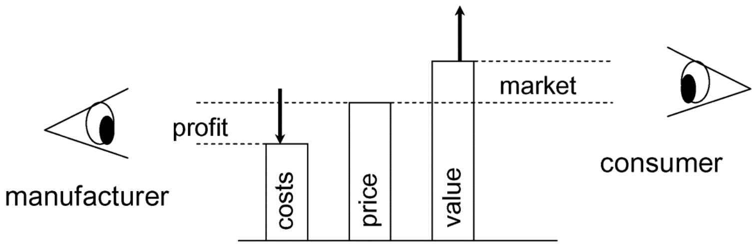


Figure A.5.2. The new management paradigm is about enhancing the value/costs ratio".

In the classical management paradigm, higher value ('quality') leads always to higher costs. In the modern management paradigm that is not the case: there are many management techniques that lead to a better value/costs ratio. Examples are: logistics (better delivery at lower stock levels), complaint management (satisfied customers with less claims), waste and quality management (less materials better quality). All these examples - there are many more in the field of Total Quality Management and Continuous Improvement – lead to more value at less costs. This is called *the double objective* for managers (Vollmann, 1996) and opens new perspectives to support eco-efficiency (it supports the first part of the eco-efficiency definition of the WBCSD). See also (Porter, 1995).
Note that this modern management philosophy is much more than just "adding services" to existing products. It is about carefully improving the quality of products and services (as perceived by the customer!) by eliminating the non-value added energy, materials and work.

The question is now whether these modern management techniques always lead to better eco-efficiency. The answer is no (e.g. the use of pesticides in agriculture results in

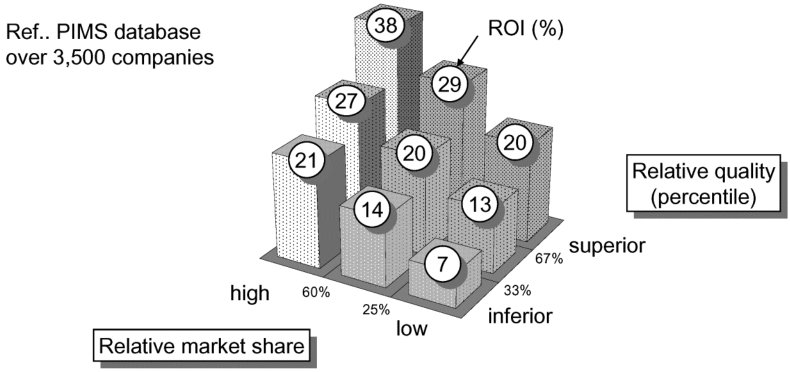
a better value/costs ratio but not in a better level of environmental protection). That is why the aforementioned definition of eco-efficiency of the WBCSD adds: "… while progressively reducing ecological impacts …".
For this reason, the "virtual eco-costs" as a single indicator for sustainability has been introduced in the EVR model of Figure 3.2. In this way the "cost structure" of a product (including services) is linked to the related ecological impacts and material depletion.

Appendix 6 [70]

The Customer Value model of Gale

General
In his book 'Managing Customer value', Bradley T. Gale has proposed a model to quantify the value of a product-service system in order to be able to analyse the competitiveness of a product portfolio of a company. This book has been written in 1994 after the PIMS study (Profit Impact of Marketing Strategy), a statistic analysis on 3500 American companies, (Buzzel, 1987). This study had revealed that the main drivers for company profits were 'relative quality' and 'relative market share', see Figure A.6.1.

Figure A.6.1. Quality and market share both drive the profitability of companies. Note that ROI is 'Return On Investment' which is the ratio of profit and invested capital



The route to a high profit is clear: via a high 'relative quality', a high 'relative market share' can be achieved, resulting in a high ROI. But the question then is how to achieve a high relative quality, being a high quality at the right price.
The key to this question is to focus on the quality dimensions (Garvin, Section 5.3) that are important to the customers (as perceived by the customers). There are two options:

1. either improve the quality/price ratio of the quality dimensions which are important to the customers

[70] The original title: Annex 5b (Vogtländer, 2001, A).

2. or try to influence the customer preferences in the direction of those quality dimensions of your own products which are relatively high in comparison with the competitors.

Option 1 is obvious: companies have to deliver products with a good quality/price ratio. Option 2 is often combined with option 1. When a car manufacturer has developed a relatively safe car, this manufacturer has to make the market aware of this fact *and has to make it an important quality dimension at the moment of purchase*. The same applies to the issue of the environment. Only the right marketing strategy will result in the desired situation that the product is perceived as better at the moment of purchase.

The model is explained here by an example, providing the main methodology, its characteristics, and the philosophy behind this model[71].
The (slightly simplified) methodology comprises three steps:
Step 1. Assessment of the 'perceived quality ratio' of the product-service system.
Step 2. Assessment of the 'perceived price ratio' and the 'fair price'.
Step 3. Assessment of the strategic consequences.

Step 1. Assessment of the 'perceived quality ratio'
Since most of the strategic marketing analyses are confidential, we will use here a hypothetical example of vegetables, where a 'bio-vegetable' (no use of pesticides) is compared with the normal vegetable.
The comparison is made by a panel of consumers, as they perceive the relative ratings (1 is the lowest score in rating, 9 is the highest score in rating). The results and the calculation scheme are given in Table A.6.1.

(1) Aspect	(2) Importance	(3) Q rating Bio-product	(4) Q rating Normal product	(5) ΔQ rating = (3) – (4)	(6) 'weighted' = (5) x (2)
Taste	0.2	8	6	+2	+0.4
Appearance	0.2	6	8	-2	-0.4
Health aspects	0.2	8	5	+3	+0.6
Presentation	0.1	7	8	-1	-0.1
Availability	0.2	6	8	-2	-0.4
Environment	0.1	8	5	+3	+0.3
Total	*1,0*	*7,1*)*	*6,7*)*		*+0.4*

Table A.6.1. Calculation scheme of the weighted quality rating of a product.

*) This is the weighted average quality = sum of quality rating × importance

The 'perceived quality ratio' is now defined as: 7.1/6.7 = 1.06
so the bio-product is rated 6 % better in terms of perceived quality.

71 The model has been linearized and slightly simplified to bring it in line with the wide spread methodology of the 'Decision Matrix'.

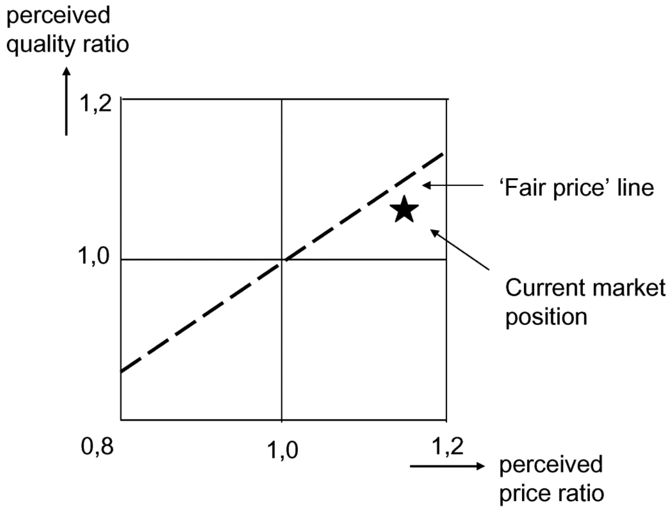
Note that this rating depends on the characteristics of the people on the panel. The Q rating as such in columns (3) and (4) do not vary much with the people on the panel. The importance of column (2), however, is very sensitive for the type of people on the panel (and therefore the so-called *market niche*). A different marketing strategy is needed for a different market niche.

Step 2. Assessment of the perceived price ratio and the fair price

In this case the perceived price ratio is simple: it is the ratio of prices for both product types. In other words, when the bio-product is 15% more expensive, the perceived price ratio is 1.15[72].

The market position of the bio-product (in relation to the normal product) is depicted in Figure A.6.2.

Figure A.6.2: The 'fair price' line and the market position of a product.



The dotted line in Figure A.6.2 is the 'fair price' line. It represents the value (in terms of money) of quality. Everything below the fair price line is perceived as too expensive; everything above the fair price line is perceived as attractive in terms of 'value for money'.

In this case, the panel stated that a maximum extra price of 10% for the bio-product was acceptable. In other words: for the majority of the people on the panel a 'perceived price ratio' of 1.1 was just acceptable (as a maximum) at the 'perceived quality ratio' of 1,06. The fair price line is then a straight line through (x = 1; y = 1) and (x = 1.1; y = 1.06).

The actual perceived price ratio (1.15) of the bio-product is then too high in comparison with the value of the product.

[72] The reason that it is called 'perceived price ratio' is that for many modern products the price is not so clear anymore (examples: mortgages, pension funds, lease contracts, service guarantees, etc.)

The first reaction of most people is that the price has to be lowered (in Figure A.6.2, the market position of the bio-product has to shift to the left). Figure A.6.2 shows, however, that there is an alternative: increase the perceived quality ratio, either by increasing the quality or by influencing the perception (change the consumer preferences).
We will analyse these options for a market strategy in Step 3.

Step 3. Assessment of the strategic consequences
In the above example, there are three options to bring the product above the fair price line:

1. lower the price by at least 5% by lowering the costs of production and distribution
2. increase the quality of the product
3. change the perception of 'what is important'.

The first option seems simple, but is often hard to realize in practice. When the sales volume is higher, distribution costs can be lower, but when a lower price does not generate the required extra volume, this option does not work. In general one should take care that savings in production and distribution does not harm the product quality (otherwise one is acting "pound foolish – penny wise"). Savings are only allowed in aspects which are not important to the customers. However, in this case there are no such opportunities.

The second option is more promising, especially when it is focussed on quality aspects which combine:

- a low score for the 'Q rating', column (3) of Table A.6.1
- a high score for 'Importance', column (2) of Table A.6.1

In this example this is the case for 'Availability' and 'Appearance'.

The third option seems to be the most attractive from the point of view of strategic marketing. The strategy here is to focus on the quality aspects with the highest scores: 'Taste', 'Health' and 'Environment'. The aim is to increase the 'Importance', column (2), as perceived by the customers. In other words, when the 'Importance' of 'Health' can be increased from 0.2 to 0.3 and the 'Importance' of 'Availability' can be decreased from 0.2 to 0.1 (the health issue becomes so important that people are prepared to buy the bio-product in a specialized shop), the scores in Table A.6.1 will change as follows:
Q rating Bio-product 7.3
Q rating Normal product 6.4

The result is that the new 'perceived quality ratio' is 7.3/6.4 = 1.14
Which is well above the fair price line.
The strategy of the three options is summarized in Figure A.6.3.

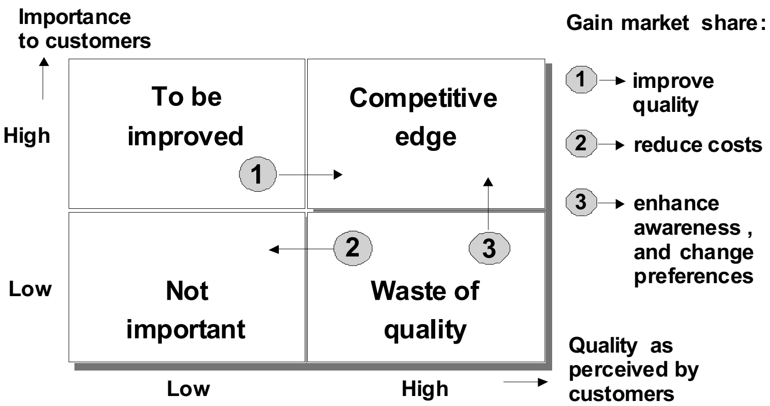


Figure A.6.3. The market strategy of products has three options in the portfolio of *Q* aspects.

Appendix 7 [73]

A method to determine the value of the 3 main aspects of Quality (the value of product quality, the value of service quality and the value of image)

Essential for the quality dimensions of Garvin (Section 5.3) is that they can only be judged by the customers ("as perceived by the customers", measured by customer panels or customer surveys). These quality dimensions can be expressed in terms of the 'fair price' for it, as described in Appendix 5 and 6.

The technique is that the customer is asked to estimate the value of the total product-service system in terms of the (total) fair price for it. The fair price is the highest price at which a customer is prepared to buy a product and/or service. When the price of a product is higher than the fair price, the product is considered by the customer as too expensive. When the price is lower than the fair price, the customer considers a purchase as attractive. See also Figures A.5.1 and A.5.2 of Appendix 5.

In addition to the assessment of the fair price, each quality dimension can be rated:

- in terms of the quality (=value) of each dimension (ranging from 'very poor' to 'excellent'; e.g. 'very poor' = 1, ' excellent' = 5)
- in terms of importance of each dimension (ranging from 'of no value' to 'very important'; e.g. 'of no value' =1, 'important' = 5).

The fair price for the product quality, the service quality and the image can be determined then by calculating the weighted averages of the ratings of the *Q*-dimensions, and assigning the corresponding portions of the total fair price to the quality aspects.
An example for an easy, linear, situation is given in Table A.7. The customer is asked to assess the total fair price (= total value), the Importance Score and the Value Rating

[73] The original title: Annex 5c (Vogtländer, 2001, A)

(bold numbers). The fair price for the value aspects is then calculated according to the scheme in the Table A.7.

	Value aspect	Importance Score (1)	Value Rating (2)	(3) = (1) x (2)	(4) = (3) / 'total (3)'	'fair price' = (4) x 'total value'
a	Product Q	**4**	**3**	12	0.40	360 €
b	Service Q	**2**	**3**	6	0.20	180 €
c	Image	**3**	**4**	12	0.40	360 €
d	*Total*			*34*	*1,00*	*900 €*

Table A.7. Calculation scheme for assessment of the fair price for value aspects (example).

Appendix 8 [74]

The three-stakeholders model

In general, individuals are neither prepared to pay more for 'green' products, nor are they prepared to give up their 'freedom' in terms of less travelling. However, most people (in The Netherlands) are quite aware of the importance of the issue of sustainability, and are aware of their responsibility in this respect[75] (Steg, 1999).
The question is now: what has to be done about it? The fact that people are positive about the issue of sustainability has to be converted to a situation where people buy sustainable products, but how?

The environmentalists seem to be more and more disappointed that the market shares for 'green products that are a bit more expensive' stay marginal (2 – 5 % in The Netherlands), irrespective of the many efforts which have been taken in the recent past (Hoefnagel, 1996, Steg, 1999, Nas, 2000, CBS, 2000).
The question is, however, whether the right measures have been taken up to now.

In this respect it is important to realize that (Senge, 1990):

- the required transition is a process rather than a quick fix;
- the system to be changed is rather characterized by complex circular interrelationships than by simple linear cause-effect relationships;
- the harder environmentalists push, the harder the existing system will push back;
- small changes in the dynamic system can produce big results – but the areas of highest leverage are often the least obvious.

In the transition towards sustainability, each of the stakeholders have to play their own role:

[74] The original title: Sections 10.1 and 10.2 (Vogtländer, 2001, A)

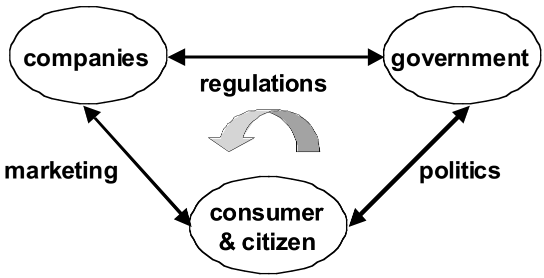
[75] A Dutch enquiry (1995) on the subject revealed a surprisingly high score on the question: "people behave irresponsibly when they do not take environmental effects into account". The average score was 4.3 on a scale of 1 (totally disagree) to 5 (totally agree).

- the consumers/citizens have to shift their expenditures towards a lower Eco-cost/Value Ratio, i.e they should buy 'green' products and services
- the companies have to create product-service combinations with a lower Eco-costs /Value Ratio, i.e. they should offer 'green' solutions to the market
- the governments have to create regulations and new systems for tax, subsidies and Tradable Emission Rights, i.e. they should create a business environment which gives 'green' solutions a fair chance in competition with the current products and services.

It is obvious that, when one of the stakeholders fails to play the right role, the transition towards sustainability will not happen. What triggers each of the stakeholders of the system to go in the right direction? Who triggers the transition process?
Designers tend to believe in 'technology push': when the green products are on the market, they will be bought in the long run, but the reality seems different.
The general business opinion is inclined to 'market pull': the consumers have to trigger off the demand. Why should they do so? In reality they tend to go for the best price/value proposition in the market instead of the proposition with the lowest environmental burden, since the latter is normally slightly more expensive. Advertising campaigns to make people buy the slightly more expensive 'green' option failed to succeed so far.
Apparently, the government should do something as well: level the playing ground in the market, i.e. create a system in which the 'green' solutions have a fair chance.

The key to the solution of the problem is to realize that the consumer is an individualist, reacting instantly and in the short term to offerings on the market. Sustainability, however, is a long term issue for the citizen. We have to realize that each individual is consumer as well as citizen.
The interactions of the consumer/citizen with companies and governments are depicted in the three stakeholders model[76] of Figure A.8.

Figure A.8. The three stakeholders and their main interactions.



The three stakeholders have three different interactions with each other:

[76] The validity of the model was checked in three computer decision room sessions with consumers, business representatives and governmental representatives. See Chapter 9 and also the questionnaire of Table 9.4.

- the citizens are interacting with their governments via politics: citizens want to have a sustainable future, are aware of the fact that the required transition can only succeed when we put our shoulders under it *together,* and therefore ask the government to take action
- the government is interacting with the companies: governments take actions via regulations, taxes, subsidies, 'convenants', etc., and force companies to react
- companies are interacting with consumers: companies try to offer consumers 'best value for money' and gain market share by satisfying the (short term) customer (individual) needs.

The predominant direction to trigger the required transitions in the circle of Figure A.8 is counter clockwise, as described above. In some business areas, industry is acting pro-actively (instead of reactively), for instance in the automotive industry. In those areas, one is trying to gain a competitive edge by being the first to meet future governmental standards, but the underlying reasoning of those business strategies is counter clockwise in terms of the three stakeholders model of Figure A.8.

Initiatives which start with the consumer business relationship (the clockwise direction) tend to remain limited to the small market niche of people (2%–6%) who regard the environmental aspects of products as more important than the price/value ratio. It seems extremely difficult to extend these markets to the 'normal' markets of price/value buyers[77].
There might be a few exceptions though in the near future, where the transition towards bigger markets for 'green' (or 'greener') products is only driven by new marketing concepts, without the direct support or intervention by governments. A most promising example is the business of the supply of 'green' electricity in the consumer markets, triggered by the internet in combination with the new opportunities in the evolving free electricity markets (Green Mountain Energy Company, USA).
Appendix 9 will deal with product portfolio strategies and marketing concepts which support such business initiatives.

[77] In some exceptional situations, pressure groups have been able to trigger consumer boycotting actions, which forced companies to shift their environmental policy (Brent Spar; some products of Sainsbury). This can happen only under special conditions (Hall, 2000), and therefore cannot be regarded as the standard road towards sustainability.

Appendix 9 [78]

Consumer marketing: the Double Filter Model and marketing of commodity products

The Double Filter Model

Consumers select the 'quality products' they buy on the basis of a comparison of the perceived value of these products. Within a certain price range they seek for the best value for money, where value can be defined as the fair price, see Appendix 6 and 7.
Consumers select the 'commodity products'(products which cannot be differentiated by quality or design) on the basis of the lowest price.

As has been mentioned in the introduction, the vast majority of the consumers (94%-98%) are not prepared to pay more for 'green' products, and 'green' products are generally slightly more expensive. In other words: the consumers are not valuing the environmental aspects of products in terms of a higher 'fair price', despite many efforts (awareness campaigns of environmentalists).

At the same time, the level of the environmental awareness in The Netherlands is quite high: 89% agrees with the statement "environmental pollution is a severe threat for our children" (Aarts, 1995). The Dutch seem even to agree on their individual responsibility in this respect.
The big issue is therefore why there is such a big difference between the environmental awareness and the behaviour of people as consumers ("consumers are short term individualists").
It seems that people feel that the environmental problems can only be resolved by a joint effort (Nas, 2000), and that individual behaviour is quite useless, *unless it is agreed (and arranged by the government) that everybody has to do it*. The Dutch proverb "change the world, start with yourself", seems to be replaced by "the government has to set and keep the rules".
In the 3 computer decision room sessions as described in Chapter 7, it appeared that the decision process to rank products in terms of personal preferences was slightly more complex than just deciding on the basis of the best value for money. Environmental aspects played a secondary role ("the Double Filter Model") as depicted in Figure A.9.1.

[78] The original title: Section 10.4 (Vogtländer, 2001, A)

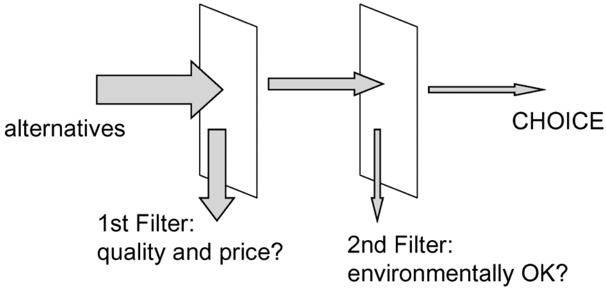


Figure A.9.1. The Double Filter Model : environmental data serve only as a second order filter in the decision of consumers.

Nearly all participants of the 3 computer decision room sessions appeared to make their first choice on the basis of value for money. However, when a decision could not be made on this first criterion, the environmental data were used to make a final choice. Apparently people make their choice first as consumer ('short term individualist'), but when that does not lead to a clear decision, they add considerations as a citizen (long term, including considerations for the society).

Marketing of commodity products and 'quality' products

Although environmental aspects seem to be a secondary criterion for the choice of consumers, it can have an enormous impact in the marketing and sales of products and services, since a high percentage of consumers care about the environment in their role as citizen.

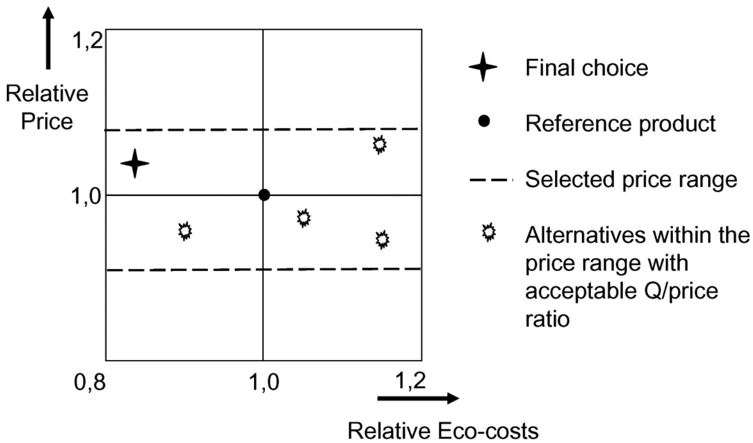
In the case of commodity products (water, electricity, food ingredients, but also electronic appliances such as video recorders and services such as travel by plane), where price is the predominant selection criterion for the market segment, the environmental aspects might become the competitive edge.

Green Mountain Energy Company in the USA was one of the first companies in the world that realizes that there is an enormous potential for 'green' electricity in a free and open energy market for consumers, *because it is hard to differentiate such a product, since all electricity is equal.* Their marketing studies in the USA show that 50% of the households are interested in green electricity. They estimate that 10 % - 20% of these households is prepared to buy green electricity, even when the price is slightly higher. This Texas based company started marketing their product in California, Pennsylvania and New Jersey where the electricity markets had been liberalized. Their approach was pure marketing driven in the modern way, appealing to the 'feel good' factor which is quite important in our current society. They didn't appeal to the responsibility in terms of sustainability (which has a bit 'heavy' and negative connotation), but just related to feeling good at holidays in mountains and forests in thier advertisments (positive things to remember)[79].

[79] The Dutch electricity distributor NUON bought 23 % shares of Green Mountain in October 2000, and started a similar marketing approach for Dutch households as of December 2000.

The basic idea of marketing green commodity products is depicted in Figure A.9.2.
Take as an example a video recorder (mono and 'show view'). The price in European discount shops is all the same, plus or minus 10%. They are all black, have the same shape and look. The choice is typically between 4 to 6 manufacturers (Sony, Philips, Panasonic, etc.) which are available on the shelf. Many people do not know which one to choose, and follow the advice of the salesman in the shop.
In those cases there is an opportunity for a secondary selection criterion: the environmental aspect.
Under the condition that the environmental burden of the alternatives is known (in terms of eco-costs or any other single indicator or equivalency), and under the condition that this is communicated in the right way, there is a high probability that the consumer will buy the greenest video recorder.
Green Mountain Energy Company was marketing its product along the same lines of reasoning: differences in price are negligible (5 – 15 € per month for a household), they offer a choice from three well explained solutions, and appeal to the 'feel good' emotions of 'reconciliation of the consumer and the citizen' within in each person.

Figure A.9.2. Eco-costs as a competitive edge in the marketing of a commodity product.



The consequence for the product portfolio management of such a marketing approach is that new environmental products have to fall within the normal price range. It is better to offer a good green solution within the price range, and sell many of them, than to go to the extreme and discover that there is no market for it.
The relevance of this marketing approach (and consumer behaviour) is not only that a small environmental gain which is applied many times is more effective than a big gain which fails the acceptance in the market, but also that every manufacturer will try to become the best in terms of sustainability. This will lead to a fierce competition on this issue. Such a competition might become a major drive for the de-linking of economy and ecology.

For quality products, the situation is more complex.

Firstly, it appeared in the 3 computer decision room sessions that the fact as to whether or not a product is green, can suddenly become the decisive factor in the final choice. This is the case when two (completely) different solutions for the same functionality are perceived as equal in terms of value for money. In such cases, the situation is comparable with the situation for commodity products and Figure A.9.1 and A.9.2 can be applied.
Secondly, it cannot be denied that, for products in the high quality range, the issue of sustainability can be made part of the product image (example: Swedish cars in comparison with American cars). The feel good factor is extremely important for the high end markets. It seems that the environmental issues have to be bundled with other societal issues (safety, health, conservation of nature and cultural heritage; in other words future welfare for our children). In this respect it is good to realize that sustainability is in the top 5 of the Dutch societal issues for the last 3 decades. A distinctive focus on the environment seems not to be a recommendable strategy: people with higher education (in The Netherlands) seem to be irritated by the extreme standpoints of environmental action groups (Nas, 2000). A careful, positive, marketing approach is required for the high end products to enhance the market share. Pushing too hard with moral statements will have an adverse effect and will hinder a gradual transition towards sustainability in consumer markets.

Appendix 10 [80]

Governmental policies for sustainability

The problem

With regard to the environment, the major task of the government is to facilitate (and enforce) the transition towards a sustainable society for their citizens. In the light of the three-stakeholders model, the role of the governments is to create regulations and new systems for tax and subsidies, in order to to create a business environment which gives 'green' solutions a fair chance in competition with the current product and services.

There are 3 important aspects with regard to governmental regulations and systems for tax and/or subsidies:

1. the commercial playing fields have to be kept level during the transition period, ensuring that a company cannot gain position by avoiding investments and innovations
2. regulations have to cope with the fact that governments can only set rules within their own territory, whereas world trade has no longer any restrictions (according to regulations within the EU, and agreements of the WTO, protection of own industries has been forbidden)

[80] The original title: Section 10.5 (Vogtländer, 2001, A)

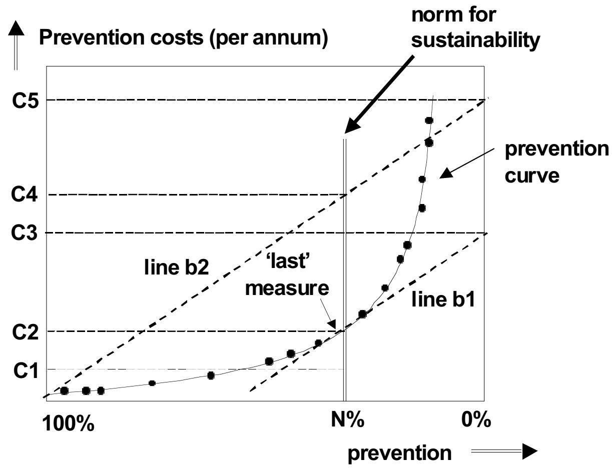
3. regulations have to stimulate innovation within the life cycle.

With regard to point 3 it is important that the transition towards sustainability will trigger innovation, which will make the transition less costly than many believe it will be. Innovations are supported by regulations in which (Porter, 1995):

1. results are regulated, not technologies (leaving maximum room for "how" the results are achieved)
2. strict requirements are set to results (leaving no room for half solutions)
3. rules are set at the end of the chain (leaving maximum room where and how to make changes in the chain)
4. free market initiatives are encouraged (to create new combinations of activities, new 'profit pools')
5. phase-in periods are defined and adhered to (innovation takes time and requires stable governmental policies).

The major problem of governmental regulations is how to combine point 1 and point 2 in an open economy. This major problem can be explained by describing the dilemmas of a few scenarios for governmental policies, analysing the effects of these scenarios by means of Figure A.10.

Figure A.10. Marginal prevention costs and total costs on the road towards sustainability.



The basics of Figure A.10 have already been explained in Appendix 2 for the case of emissions:

For each type of emission, the costs and the effects (in terms of less emissions) are accumulated for several prevention measures to be taken (a 'what if' calculation), the 'prevention curve'. At a certain point on the curve, the 'norm for sustainability' is reached. The marginal prevention costs are defined by the costs per kg reduction of the 'last' measure, depicted as the slope of line b1.

The basic problem is *not* that the total costs to reach the norm for sustainability are too high. For acidification + eutrophication + summer smog + fine dust + eco-toxicity, the total Dutch prevention costs per annum, C2, of the curves as calculated in Chapter 2 are approximately 2.5 billion €, being only about 0.7% of the Dutch Gross National Product (GNP). The annual costs of prevention measures for CO_2, as proposed in Chapter 2, are estimated at 2.3 billion € (including savings of energy consumption) to reach the Kyoto norm, being approx. 0.6% of the Dutch GNP [81].
The problem is though how to distribute the costs of those measures: who pays what? Since those measures are distributed over all sectors of our economy, and since the price of each measure is different in terms of €/kg, it is not feasible to 'dictate and specify' technical measures to all parties involved. So what is to be done in terms of regulations to lead the market towards sustainability? Taxation? Tradable emission rights? Subsidies?
The problem will be analysed by separate scenarios for tax, tradable emission rights, and subsidies.

Taxation
The simplest way of pushing every party involved towards sustainability is taxation:
Suppose that emissions are taxed at the level of the marginal prevention costs (€/kg) at the norm for sustainability (this cost level is depicted in Figure A.10 as the slope of line b1).
The result will be that all parties will take prevention measures rather than pay tax, if these measures are at the left side of the norm for sustainability in Figure A.10. Parties at the right side of the norm will prefer to pay the tax, since the measures are more expensive.
So taxation is effective and simple in terms of pushing all parties involved in the right direction.

The disadvantage of such a sudden, simple taxation system is that the total sum of tax for emissions is exorbitantly high at the start of the transition: see point C5 in Figure A.10. This high tax level will be rather devastating for our economy, so citizens are not likely to make such a heavy sacrifice for sustainability.
After the prevention measures have been taken, the total costs of measures + tax will become lower: point C3 in Figure A.10.

Increasing the tax in small increments over a long period of time will resolve the problem of the wild disruption of our economy (point C5), but the end result will be that the sum of measures + tax is still rather high (point C3). Especially in an open economy high cost levels for emissions are problematic in terms of the international

[81] The required extra investments in prevention measures for acidification + eutrophication + summer smog + fine dust + eco-toxicity are estimated at 25–30 billion €. The required investments in prevention measures for CO2 are estimated at approx. 20 billion € (Beeldman, 1998) to reach 50% reduction of the level 1990.

trade: products become too expensive to stay competitive and there is a big likelihood that production will be moved outside the borders ('export' of environmental pollution).

One can conclude that tax may be effective from the point of view of reduction of pollution within a country, but is not advisable from an economic point of view. It has a much bigger influence on the economy than seems to be required: the method seems to be to coarse (the tax which is paid at the right side of the norm in Figure A.10 does not serve any purpose).

Tradable Emission Rights

Systems for Tradable Emission Rights (TER) are much more subtle that tax systems. Those systems support the flow of money to investments on the most cost effective prevention measures in the industry (Sorrell, 1999). TER systems can only be applied in combination with rules which restrict 'free' emissions of production plants.

The first scenario is the most pure form of TER:

- free emission rights of production plants are restricted to the current emission levels (so the current level of emissions are the maximum free emission rights for the future)
- companies are allowed to buy emission rights from other companies.

The result of such a system is twofold:

- companies which expand, have to buy emission rights from other companies to extend their allowable emission levels
- companies which have the opportunity to reduce their emissions by prevention measures, will do so when they can sell their emission rights at a higher price than their own investments required for prevention.

In such a system the market will take care of an optimal distribution of prevention measures in terms of cost effectiveness.

When total emission levels have to be reduced, the situation becomes slightly more complex. Emission reductions of 10%, 20% or 30% will work basically in the same way along the lines of the aforementioned scenario. The reductions have to be implemented gradually (in small steps and slowly) to avoid heavy disruptions to the TER prices[82].
Reduction of a factor 3 or 4, as depicted in Figure A.10, can better be implemented when the government takes part in the trading system. The first scenario has then to be replaced by another scenario, which is summarized hereafter.

[82] The dynamic behaviour of such a "volume controlled" system results in unstable prices because of the lead time of the prevention measures (control loops with a lead time are inherently unstable).

This second scenario is then as follows:

- the government sets free emission levels to 1/3 or 1/4 of the current level (these TERs are given at no costs),
- the government sells the additional TERs at a *levy per annum* at the price level of *x*% of the marginal prevention costs per annum; x is slowly increased, step by step, until 100% is reached[83],
- companies can give the TERs back to the government (to avoid the annual levy), which they are inclined to do when their own prevention measures (in terms of costs per annum) are less expensive,
- companies can sell TERs to other companies as well (at a price level of the annual levy, or, at a slightly higher price per annum).

The advantage of such a scenario is:

- the burden of the levies is gradually increasing, investments to avoid the levy can be carefully scheduled,
- companies are free to expand their activities, at the cost, however, of the price of the TERs they can buy from other companies,
- the total burden at the moment of introduction can stay rather low, for instance at the level of C1 (lower than the total prevention costs, C2, to minimize the economic disruption at the start),
- the total burden will rise to a maximum level of C2 at the end of the transition.

Note that governments might compensate the levies at the beginning of the transition with a tax relief (shift the burden of tax), as long as international agreements against protection of national industries are not violated.
Note also that the basic problem of introducing TER systems is the assessment of the level of applied technologies in current industries: companies which already did a lot in the field of emission prevention should not be 'punished' at the introduction by getting the emission rights at the same price as companies which did nothing in the past. At the moment of introduction, the emission level per output quantity and the 'distance' to BAT (Best Available Technology) has to be brought into the equation.

An important advantage of TER systems is that these systems relate the environmental measures to the direct product costs in a way which is clear to all stakeholders (eco-costs become real costs in a direct and understandable way). When the TER costs in a product are directly communicated to consumers, consumers with a high level of environmental awareness are able to avoid products with a high level of TER costs (resulting in a shift towards a lower EVR).

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[83] Note that, when the TERs are introduced at the level of the marginal prevention costs at the norm, this would lead to a costs burden C4 at the moment of introduction.

Subsidies

Subsidies are not suitable for industry: they disturb the inherent competitiveness of companies and they are normally violating regulations of international trade systems (it protects the national industry).

On the other hand, subsidies are suitable on the level of households. Here they can be applied as well in the form of tax relief.

The aim of subsidies is to influence the expenditures, especially on investments.

Subsidies have to be applied with great care:

- subsidies have to be specific for market niches (see the example of the TV of Chapter 5)
- the rebound effect has to be taken into account (generally speaking, subsidies can only be applied to products with a high EVR, see Chapter 5).

Subsidies are the ideal tool to support market introductions of innovation: the new sustainable solution can be made less expensive for the consumer than the classic product (for the period of the learning curve and the economies of scale curve).

A good example of tax relief on products was the introduction of the catalyst gas exhaust pipe of cars. It is likely that innovative motor designs such as the fuel cell motor need to be supported by subsidies as well to bridge the first production period with a low 'economies of scale'.

Innovation of fuels for cars (such as ethanol from biomass) might also be supported by tax relief systems (as has been done in the past for lead free gasoline).

Postscript

On Venice … …

Venice is a typical example of how a downward spiral of poverty and environmental degradation can be turned around into an upward spiral of more wealth and less pollution.
The local economic activities were shifted from activities with a high EVR (chemical industry) towards activities with a low EVR (the service industry).

The biggest local threat for Venice now is the increasing frequency of flooding. This problem is related to a global cause: the climate change. This problem, however, has a local answer as well: construction of a seawater barrier which is normally open, but which can be closed at high tides.
It is rather astonishing that local environmental action groups are opposing such a technical measure. They seem to stick to their fundamentalist dogmas, rather than do something about this major sustainability issue for the city!

On Diogenes … …

Contemplation is good for mental health in all times: in the past and in the future.
However, with regard to sustainability, the "Diogenes solution" is not the right answer for our future world. To provide sufficient living conditions for so many people in our world, sound economic systems will be required, systems with a high eco-efficiency.

The right solutions of the past apparently differ from the right solutions for the future.

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Summary

The Eco-costs/Value Ratio, EVR

The basic idea of the EVR (Eco-costs/Value Ratio) model is to link the 'value chain' (Porter, 1985) to the ecological 'product chain'. In the value chain, the added value (in terms of money) and the added costs are determined for each step of the product "from cradle to grave". Similarly, the ecological impact of each step in the product chain is expressed in terms of money, the so-called eco-costs. See Figure A.

Value :	value	+	Δ value	+	Δ value	+	Δ value	+	Δ value	+	Δ value	=	**Total value**
Costs :	costs	+	costs	+	costs	+	costs	+	costs	+	costs	=	**Total costs**
Eco-costs	eco-costs	+	eco-costs	+	eco-costs	+	eco-costs	+	eco-costs	+	eco-costs	=	**Total eco-costs**

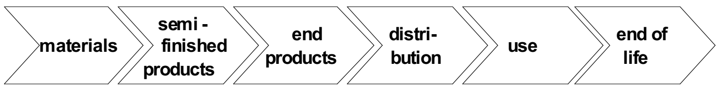


Figure A. The basic idea of combining the economic and ecological chain: "the EVR chain".

The eco-costs are 'virtual' costs: these costs are related to measures which have to be taken to make (and recycle) a product "in line with earth's estimated carrying capacity". These costs have been estimated on the basis of technical measures to prevent pollution and resource depletion to a level which is sufficient to make our society sustainable.

Since our society is yet far from sustainable, the eco-costs are 'virtual': they have been estimated on a 'what if' basis. They are not yet fully integrated in the current costs of the product chain (the current Life Cycle Costs). The might be regarded as 'hidden obligations'.

The ratio of eco-costs and value, the so-called Eco-costs/Value Ratio, EVR, is defined in each step in the chain as:

$$\text{EVR} = \text{eco-costs} / \text{value}$$

For one step in the production+distribution chain, the eco-costs, the costs and the value are depicted in Figure B.

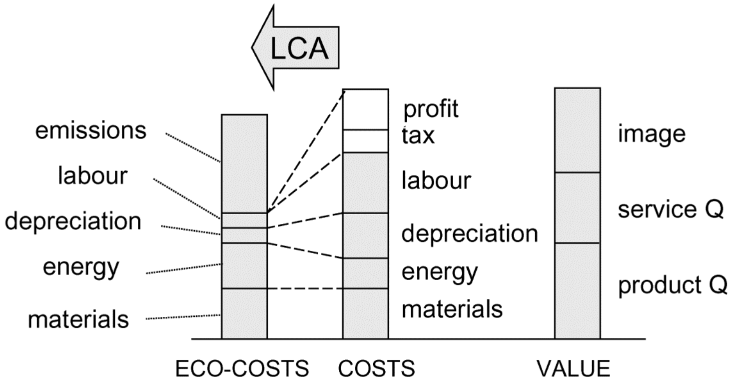


Figure B. The decomposition of "virtual eco-costs", costs and value of a product.

The five components of the eco-costs have been defined as 3 'direct' components plus 2 'indirect' components:

1 virtual pollution prevention costs, being the costs required to reduce the emissions of the production processes to a sustainable level (Chapter 2)
2 eco-costs of energy, being the extra price for renewable energy sources
3 materials depletion costs, being (eco-costs of raw materials)×(1 – α), where α is the recycled fraction
4 eco-costs of depreciation, being the eco-costs related to the use of equipment, buildings, etc.
5 eco-costs of labour, being the eco-costs related to labour, such as commuting and the use of the office (building, heating, lighting, electricity for computers, paper, office products, etc.).

Based on a detailed cost-structure of the product, the eco-costs can be calculated by multiplying each cost element with its specific Eco-costs/Value Ratio. These specific EVRs have been calculated on the bases of LCAs. Tables are provided for materials, energy and industrial activities. See www.ecocostsvalue.com tab data.
(See Chapter 3.)

The pollution prevention costs

The aforementioned pollution prevention costs are being calculated in four steps:

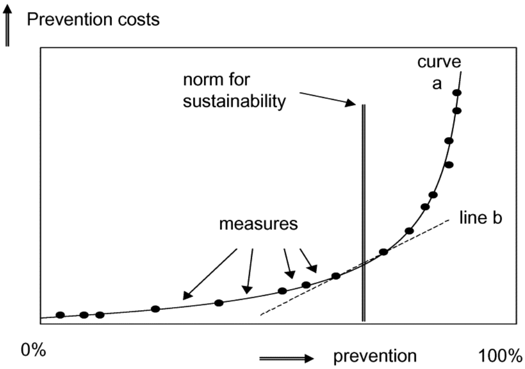
1. LCA calculation according to the current standards (ISO 14040 and 14044)
2. Classification of the emissions in 7 classes of pollution
3. Characterization according to characterization multipliers as used in e.g. the Eco-indicator '95, resulting in "equivalent kilograms" per class of pollution
4. Multiplication of the data of step 3 with the 'prevention costs at the norm' ,being the marginal costs per kilogram of bringing back the pollution to a level "in line with earth's carrying capacity".

The following 'prevention costs at the norm' are proposed for The Netherlands and Europe:

1	prevention of acidification	7.55 €/kg SOx equivalent
2	prevention of eutrophication	3.60 €/kg phosphate equivalent
3	prevention of ecotoxicity (heavy metals)	802 €/kg Zn equivalent
4	prevention of carcinogens	33 €/kg PAH equivalent
5	prevention of summer smog	8.90 €/kg C_2H_4 equivalent
6	prevention of fine dust (winter smog)	27.44 €/kg fine dust PM2.5
7	prevention of global warming	0.135 €/kg CO_2 equivalent.

These 'prevention costs at the norm' are based on the so-called 'marginal prevention costs' of emissions. The way these marginal prevention costs are determined is depicted in Figure C. For each type of emission, the costs and the effects (in terms of less emissions) are accumulated for several prevention measures to be taken (a 'what if' calculation). At a certain point on the curve, the 'norm for sustainability' is reached. The marginal prevention costs are defined by the costs per kg reduction of the 'last' measure, depicted as line b.
The 'norms for sustainability' are based on the 'negligible risk levels' for concentrations (in air and in water).

Figure C. The way the marginal prevention costs are calculated from emission prevention measures for a certain region.



(See Chapter 2)

The End of Life stage and recycling: Cradle to Cradle

The End of Life systems are rather complex. For complex products, like buildings, there are many different system opportunities to make the solution more sustainable (from recycling to enhancement of the durability).
Figure D depicts the major types of End of Life treatment and types of recycling. It is developed to describe and analyse the various kinds of complex modern life cycles of consumer products, buildings, manufacturing plants, civil structures, etc.

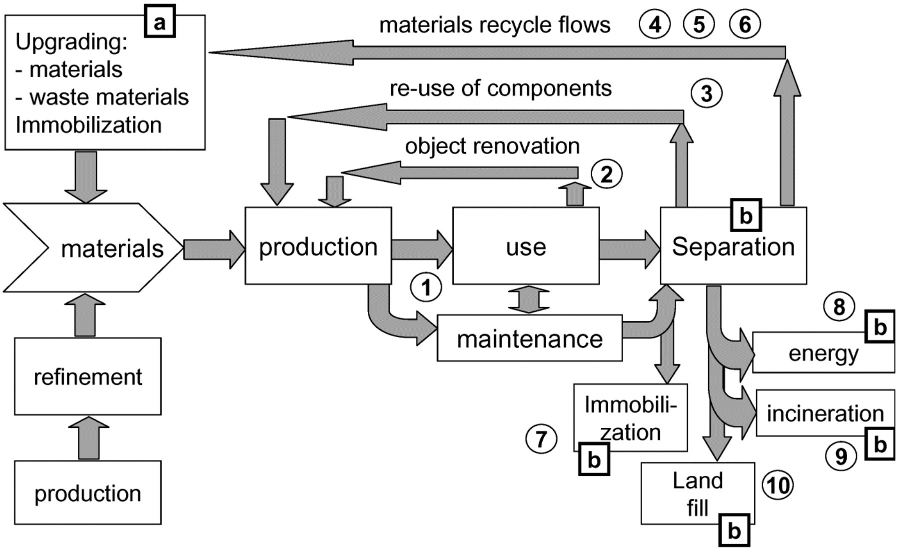


Figure D. The flow of materials in the Life Cycle.

The numbers in Figure D relate to the "Delft Order of Preferences"[84], a list of the 10 major systems for End of Life, used for structured and systemized analyses of (combinations of) design options:

1. Extending of the product life
2. Object renovation
3. Re-use of components
4. Re-use of materials
5. Useful application of waste materials (compost, granulated stone and concrete, slag, etc.)
6. Incineration with energy recovery
7. Immobilization with useful appliances
8. Incineration without energy recovery
9. Immobilization without useful appliances
10. Land fill.

It is important to realize that for big, modular objects (like buildings), there is not "one system for End of Life" but in reality there is always a combination of systems.
Two basic rules for allocation in the EVR model are:

1 Costs and eco-costs of all activities marked with 'b' are allocated to the End of Life stage of a product (transportation included).

2 Costs and eco-costs of all activities in the block marked with 'a' are allocated to the material use of the new product (so are allocated to the beginning of the product chain).

[84] In 2007 the order of numbers 6-9 was slightly changed (incineration was placed higher than immobilization, because of the positive effects in LCA calculations)

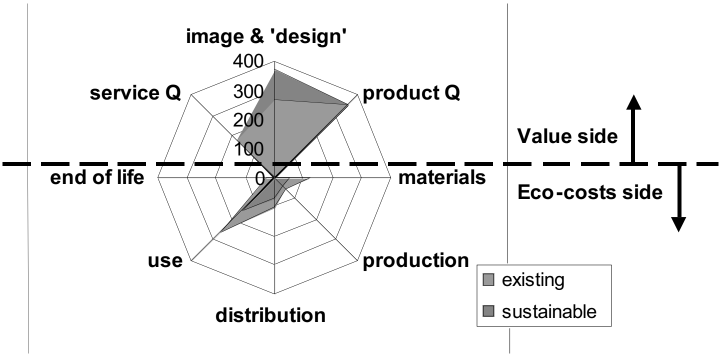
In line with the aforementioned allocation strategy, the 'bonus' to use recycled materials is taken at the beginning of the product chain, where the new product is created. Material depletion is caused here when 'virgin' materials are applied, material depletion is suppressed when recycled materials are applied.
(See Chapter 4)

Ecoefficient value creation

Product designs for the future will need to combine a high value/costs ratio as well as a high eco-efficiency.
The advantage of the EVR model is that it can reveal how the de-linking of economy and ecology can take place in practical situations.

For designers, the EV Wheel has been developed, showing the strength and weakness of a certain design on the value side as well as the eco-costs side. See Figure E. A sustainable design is characterized by high scores at the value side and low scores at the eco-costs side.

Figure E. The Eco-costs & Value Wheel (EV Wheel), with value and eco-costs (€).



Another powerful instrument to analyse a product is an eco-costs value chart of the manufacturing, assembly and distribution chain.
In the production chain, the value as well as the eco-costs gradually increase from the raw materials to the point of sales. This is depicted in the example of a 28" television in Figure F.
The EVR is also a good indicator of the sustainability of consumers expenditures. The so-called "rebound effect" is depicted in Figure G, showing that 'savings' are sometimes not a good solution for sustainability.
When eco-costs are reduced by 'savings', the economic value (costs for the consumer) is reduced as well, so the consumer will spend the money somewhere else. In the example of product 1 of Figure G, the net result is positive, since the money which is saved, is spent on another product with a lower EVR. In the example of product 2 of Figure G, however, the net result is negative, since the saved money is spent on a

product with a higher EVR.

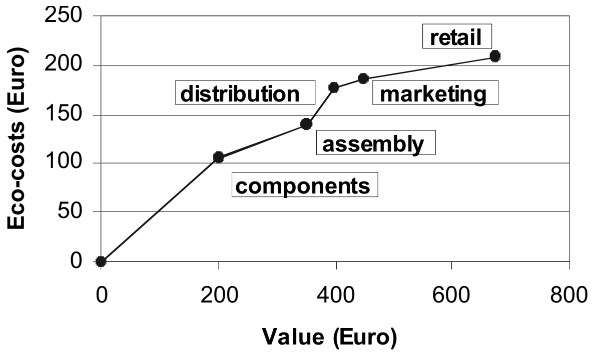


Figure F. The value and the eco-costs cumulative along the production and distribution chain (data for a 28" television).

The conclusion is that "savings" are only positive for the environment when savings are achieved in areas with a high EVR (and spent in areas with a low EVR).

A typical example of the rebound effect is related to the efficiency increase of light bulbs: when consumers spend the saved energy on more light (e.g. in their gardens) or on electricity for other domestic appliances, it does not help much in terms of sustainability.

In general, however, one may conclude that savings on energy can have a positive effect in terms of sustainability, since the EVR of energy is relatively high in comparison with other expenditures.
Savings on luxury goods (generally a low EVR because of the high labour content, might be negative for the environment since the "rebound" might be in the area of more energy (in the form of travel).

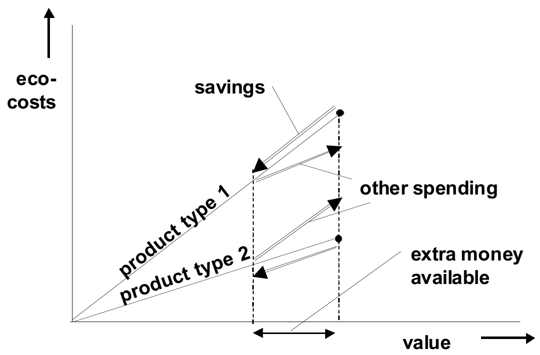


Figure G. The "rebound effect" of consumer expenditures.

(See Chapter 5)

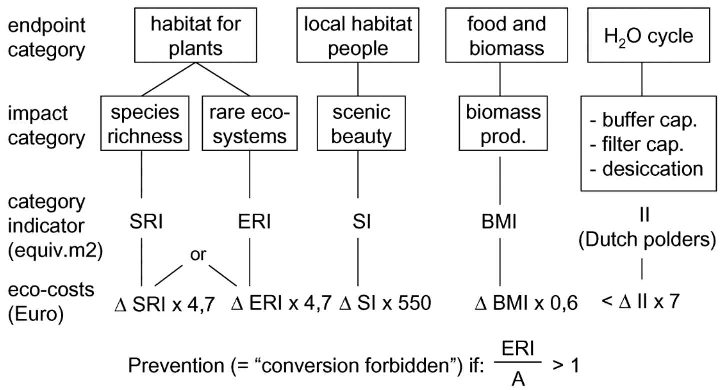
The eco-costs of land-use

Although it is argued that land-use in general cannot be integrated in the LCA of industrial mass products, a characterization system has been developed for conversion of land, since it is an indispensible element in special cases, such as:

1 LCAs of one-off products like buildings, roads etc.
2 rural and urban planning, to determine the best option and to assess possibilities of compensation (e.g. in the Dutch MER system).
3 For the eco-costs of tropical hardwood in LCA (taking into account the loss of biodiversity in tropical rain forests).

The calculation system for the eco-costs of land conversion is summarized in Figure I. The characterisation factors for species richness (as an indicator for loss of biodiversity) are use in practice, since the required data are known all over the world. For The Netherlands it is better to apply the available data for rare eco-systems.

Figure I. Characterization system for land conversion, and the corresponding eco-costs.



(See Chapter 6)

Communication

It has been tested whether or not the EVR model leads to a good understanding of the eco-efficiency of a product-service combination. In an experiment, 3 separate groups of 8-11 people were asked to rank four alternative solutions of a product-service system (the after sales service and the maintenance service of an induction plate cooker) in terms of sustainability. The 3 respective groups were:

1 customers (among whom representatives of consumer organizations)
2 business representatives from the manufacturing company of the induction plate cookers
3 governmental representatives (employees of the Dutch ministries of environmental

affairs and economic affairs, and of the Dutch provinces as well as consultants involved in governmental policies), all experts in the field of sustainability.

The instruction was to rank the proposed alternatives in terms of 'best sustainability ' as well as in terms of 'best choice in general', and to give arguments for the chosen answers. Furthermore it was asked what information was missing to make 'the right' decision on the ranking (as it was perceived by the participants).
At the end it was asked whether the eco-costs and the EVR were perceived as good criteria on which to base decisions.

From the experiments it can be concluded that:

1 The concept of Eco-costs was accepted by the majority of the non-experts, in preference to LCA output on which to base their ranking
2 The concept of the EVR was understood by the majority of the non-experts, but the consequences of it in terms of life style were not easily accepted (in particular the consumers group rejected the idea to judge on their life style by an eco-efficiency parameter)
3 The environmental experts in the governmental group did not directly accept the concept of eco-costs model (they wanted in-depth information first); they tended to stick to their existing knowledge of LCA data, which is in line with Rogers' theory of diffusion of innovation.

The experiment indicates further that:

1 the aspect of sustainability plays hardly any role in the decision when a consumer has a strong preference (based on other aspects, like the cost/benefit ratio) for a certain product type
2 however the aspect of sustainability can play a quite important role in the decision when there is no preference on other grounds.

This way of selection of products and services is depicted in Figure J

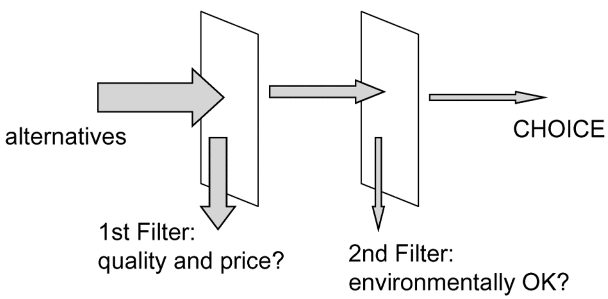


Figure J. The Double Filter Model : environmental data serve only as a second order filter in the decision of consumers

(See Chapter 7)

Case: the transport function

To illustrate the advantage of the EVR model in cases where service plays an important role, a transport chain has been analysed: vegetables from the Dutch greenhouse to the retail shops in Frankfurt. The chain has been analysed for three transport packaging systems: 'one way' solid board boxes, returnable foldable cratesand (20 round trips) returnable rigid crates (30 round trips). See Figure H.

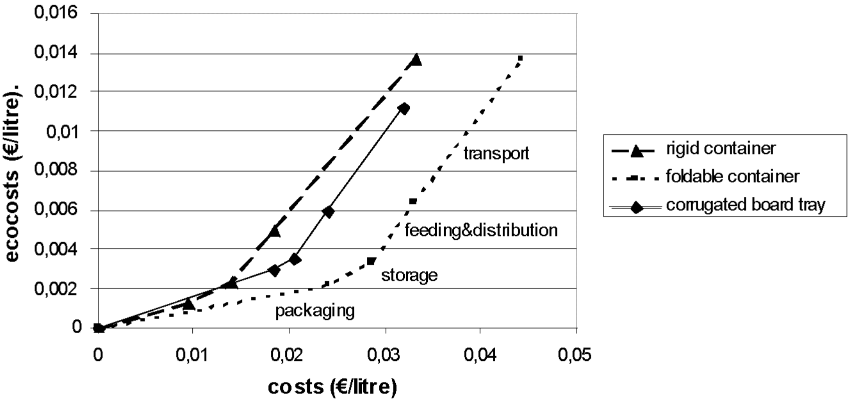


Figure H. Costs and eco-costs per litre net transport volume for the total chain (total distance 500 km); price levels 2004.

For the design of transport systems, an integral LCA approach of the total transport chain (cycle) is required to minimize eco-costs. This is because of the high interaction of the system components: the packaging system, the transport system and the storage system.

Efficient use of volume (of the truck as well as of the transport packaging) plays a key role as well as the re-use of packaging materials.

The eco-costs for the solid board box system appeares to be lower for distances lower than 200 km, especially when the truck can be used for other freight on the return trip. Costs are lower at distances more than 500 km So there is no reason from the environmental perspective to prefer plastic re-usable crates, which is an embarrassing conclusion in the light of the discussions in The Netherlands that started in the early nineties: 'durable' does not go hand in hand with 'sustainable' in this case, because the use of energy appears to be rather dominant in those transport systems.

(See Chapter 8)

Cases: Recycling of building materials

The environment is an important subject in the construction sector. This is why the following for cases have been further analysed, using the EVR model:

- What is the environmental advantage in replacing gravel in concrete with concrete aggregate?

- Can the required sand extraction on land be replaced with sand extraction at sea?
- From an environmental point of view, is it better to use mixed aggregate in concrete than in roads?
- What is the environmental advantage of a mobile crusher as opposed to a static crusher?

Analysis of this leads us to the following conclusions regarding the environment:
1. The advantage of using concrete aggregate (rather than gravel) for concrete lies primarily in the reduced amount of material dumped. Differences in emission levels are negligible
2. From an environmental point of view, sand extraction at sea is not preferable to sand extraction on land
3. Although two totally different systems are used, in the end there is little difference between using concrete aggregate in concrete and using mixed aggregate in roads.
4. From environmental point of view, a mobile crusher is preferable to a static crusher.

(See Chapter 9)

The road towards sustainability

The combined approach of eco-costs and value reveals new opportunities to reach at least "factor 4" in eco-efficiency. The required transformation, however, is far from easy.
In order to describe the mechanism of the required transition, the 'three-stakeholders model' has been introduced. See Figure K. This model provides the main interactions between business, government and consumers/citizens with regard to the issue of sustainability:
- citizens ask the government to care for their long term interest and to create sustainability
- the government defines restrictive rules and has to create an even playing field for the industry
- the industry satisfies short term consumer needs in terms of maximum value for money.

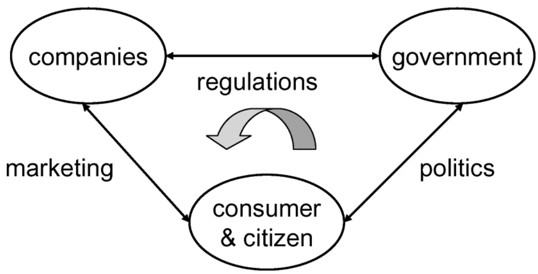


Figure K. The 'three-stakeholders model' and their main interactions.

With regard to the introduction of green products, the EVR model reveals two important issues:

1 in the product portfolio management strategy, companies have to enhance the EVR of products with a high value/costs ratio (rather than try to enhance their cost/value ratio of products with a low level of eco-costs, as many environmentalists propose). See Figure L.

2 marketing strategies need to be differentiated:
   - for commodity products (products where it is hard to differentiate on price/value) the low eco-costs of a product create a competitive edge, but keep the price/valueatthesamelevel
   - make the eco-costs part of the image for special products and high quality products, but do not stress the sustainability issues too much, since consumers go for the best price/value.

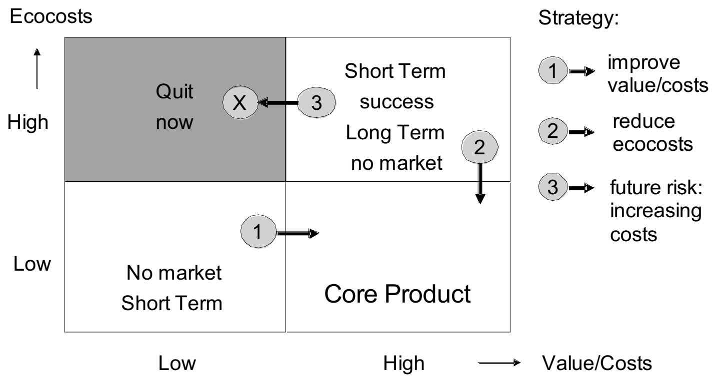


Figure L. Product portfolio matrix for product strategy of companies.

It is shown why it is so difficult for governments to force the industry in the direction of sustainability, and keep an even competitive playing field at the same time. Gradually increasing tax on pollution would work in a closed economy, but has the adverse effect of 'exporting environmental pollution' in an open, global, trade.

Tradable Emission Rights systems for the industry, in which the government takes part, seem to be the most promising solution at national level.

On global level a Tradable Emission Rights system between governments might become the right tool to freeze the CO_2 emissions at its current level. Drastic and fast reduction of the emissions, however, cannot be expected from such a system. The goal can only be reached step by step.

Systems of subsidies (or tax relief) on consumer products are suitable to facilitate the market introductions of innovative products, but only in market niches, and only for products with a high EVR. General subsidies (or tax relief) for other than these products have to be avoided.

(See Appendices)

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Sustainable Design Series of Delft University of Technology

Eco-efficient value creation

From: 'our common future', G.H. Brundtland: 'The downward spiral of poverty and environmental degradation is a waste of opportunities and of resources. In particular it is a waste of human resources. These links between poverty, inequality, and environmental degradation formed a major theme in our analysis and recommendations. What is needed now is a new era of economic growth - growth that is forceful and at the same time socially and environmentally sustainable.'

The key to such a better economy is the development of products and services which create more value and have a better eco-efficiency as well.

The model of the Eco-costs/Value Ratio is an indispensable decision support tool for architects, designers, engineers and business managers. The model is LCA based, and enables cradle to cradle design. The model features an innovative approach to the issue of a single indicator in LCA, being the so called eco-costs. The calculation is transparent and relative simple and requires no weighting steps. Since the eco-costs system is a monetary system, comparisons and analyses are possible in combination of the market value (the fair price).

This book is a compilation of the original publications on the subject. For the convenience of the reader, all tables and other data have been updated with the new eco-costs 2007 dataset.

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