Sustainable Waste Management
– methods and framework for analysis

PhD in Industrial Ecology - Dissertation

Anders G. Klang

Norwegian University of Science and Technology
Faculty of Engineering Sciences and Technology
Department of Hydraulic and Environmental Engineering

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Abstract

SUSTAINABLE WASTE MANAGEMENT- Methods and framework for analysis
ISBN 82-471-6894-4
ANDERS KLANG
MID SWEDEN UNIVERSITY
SE-831 25 ÖSTERSUND
+46-63-165760
anders.klang@miun.se

Dealing with waste is a major issue in our endeavour to create a sustainable society. The purpose of this thesis is to develop a model for assessing sustainable development in waste management systems. The model should provide a valid, reliable, useful and efficient assessment tool for waste management planners. The objective has been to improve methodologies that can support decision-making processes for sustainable waste management.

The study is based on a series of case studies in which suitable indicators of sustainability aspects for different types of waste have been developed and incorporated into a model that can be used to assess the potential for sustainable development of waste management systems. Some important methodological conclusions can be drawn:

1. Municipal decision-makers prioritise both environmental and financial issues as well as social aspects when they evaluate waste management methods (Paper IV).
2. Municipal decision-makers and municipal officers consider a holistic systems approach preferable to one that is based only on costs or only on the environmental effects (Paper IV).
3. A method that includes social aspects as well as economic and environmental ones might reach different conclusions than one that only considers eco-efficiency (Paper I).
4. A systems analysis is very dependent on definitions of the complementary systems, so a thorough understanding of possible technological development is crucial for the reliability of the analysis (Paper II).
5. Methods generating results on a very aggregated level, without employing a full life-cycle perspective, or omitting one or more aspects of sustainability, are not valid for sustainable waste management decision making (Papers I, III and IV)
6. A wide-spread opinion among local decision makers and municipal officers, that national directives and objectives for waste management are poorly adapted to the conditions of sparsely populated regions (Paper IV), does not seem to find support from systems analysis using sustainable development indicators (Papers I and II).

In addition to the methodological findings, the study also reveals how different options for waste management in a sparsely populated Scandinavian region perform differently in relation to a sustainable development. The study includes solid municipal waste, wastewater and demolition waste.

KEYWORDS: Sustainable development, waste management, decision-making, indicators.

Sammendrag

BÆREKRAFTIG AVFALLSHÅNDTERING – Metoder og analysrammeverk
ISBN 82-471-6894-4
ANDERS KLANG
MID SWEDEN UNIVERSITY
SE-831 25 ÖSTERSUND
+46-63-165760
anders.klang@miun.se

Håndtering av avfall er en hovedoppgave i vår anstrengelse for å skape et bærekraftig samfunn. Hensikten med denne avhandlingen er å utvikle en modell for å vurdere bærekraftforhold i systemer for håndtering av avfall. Denne modellen skulle tilby et gyldig, pålitelig, nyttig og effektivt verktøy for vurderinger hos planleggere innen avfallssektoren. Målet med arbeidet har vært å forbedre metodikkene som kan støtte opp om beslutningsprosesser for bærekraftig avfallshåndtering. Arbeidet er basert på en serie casestudier, der egnede indikatorer for bærekraftaspekter, for ulike typer av avfall, har blitt utviklet og innkorporert i en modell som kan brukes til å vurdere mulighetene for bærekraftig utvikling av systemer for håndtering av avfall. Noen viktige metodiske konklusjoner kan trekkes:

1. Kommunale beslutningstakere prioriterer både miljømessige og økonomiske forhold, så vel som sosiale aspekter, når de bedømmer metoder innen avfallshåndtering (Artikkel IV).
2. Kommunale beslutningstakere og tjenestemenn foretrekker en holistisk systemtilnærming i forhold til en tilnærming som kun er basert på kostnad eller miljømessige forhold (Artikkel IV).
3. En metode som inkluderer sosiale aspekter så vel som økonomiske og miljømessige vil kunne lede til ulike konklusjoner enn en metode som bare vurderer øko-effektivitet (Artikkel I).
4. En systemanalyse er svært avhengig av definisjoner i det komplementære system, slik at en grundig forståelse av mulig teknologisk utvikling er kritisk for påliteligheten av analysen (Artikkel II).
5. Metoder som genererer resultater på et meget aggregert nivå, uten om basere seg på et fullstendig livsløpsperspektiv, eller som utelater et eller flere aspekter ved bærekraft, er ikke gyldige for beslutninger med sikte på bærekraftig håndtering av avfall (Artikkel I, III og IV).
6. En utbredt oppfatning blant lokale beslutningstakere og kommunale tjenestemenn, at nasjonale direktiver og mål for avfallshåndtering er dårlig tilpasset betingelsene i tynt befolkede regioner (Artikkel IV), synes ikke å finne støtte fra systemanalyse med bruk av indikatorer for bærekraftig utvikling (Artikkel I og II).

I tillegg til de metodiske funn avdekker arbeidet også hvordan ulike mulige løsninger innen avfallshåndtering i tynt befolkede Skandinaviske regioner presterer forskjellig i relasjon til en bærekraftig utvikling. Studien inkluderer kommunalt fast avfall, avløpsvann og riveavfall.

NØKKELORD: Bærekraftig utvikling, avfallshåndtering, beslutningstaking, indikatorer.

Preface

This thesis is the result of a PhD-project within the Interreg-project ‘Gränslös Kunskap’ (Boundless Knowledge), co-organised by Mid-Sweden University in Östersund and the Norwegian University of Science and Technology in Trondheim. The thesis is based on work that has previously been presented at scientific conferences and in international scientific journals. Four papers form the theoretical and empirical basis of the thesis. I am the principal author of three of them. In Paper III, written by Grönlund, Klang et al, 2004, my work mainly consists of the sustainability assessment using the four socio-ecological principles, and contributions to the discussion. My supervisors have been Professor Helge Brattebø, NTNU and Dr Per-Åke Vikman, Mid-Sweden University.

Östersund, January 13th, 2005

Anders Klang

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List of publications

This doctoral thesis is based on the following papers:

I. Klang, A; Vikman, P-Å; Brattebø, H. (2003). Sustainable management of demolition wastes – an integrated model for the evaluation of environmental, economic and social aspects. *Resources Conservation and Recycling*. Vol 38, Iss 4, pp 317-334

II. Klang, A; Vikman, P-Å; Brattebø, H. (2004). Sustainable management of combustible household wastes – expanding the integrated evaluation model. Submitted to *Resources, Conservation and Recycling*.

III. Grönlund, E; Klang, A; S. Falk and J. Hanaeus (2004). Sustainability of waste water treatment with microalgae in cold climate, evaluated with emergy and socio-ecological principles. *Ecological Engineering*. Vol 22, Iss 3, pp 155-174.

IV. Klang, A; Vikman, P-Å; Brattebø, H. (2004). Systems analysis as support for decision-making towards sustainable municipal waste management – a case study. Submitted to *Waste Management & Research*.

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LIST OF ABBREVATIONS

CERA:	Cumulative Energy Requirements Analysis
EIA:	Environmental Impact Assessment
EMS:	Environmental Management Systems
EPR:	Extended Producer Responsibility
EPS:	Environmental Priority Strategies in Product Design
ERA:	Environmental Risk Assessment
ETC/W:	European Topic Centre on Waste
EWC:	European Waste Catalogue
GDP:	Gross Domestic Product
LCA:	Life Cycle Assessment
MFA:	Material Flow Accounting
MIPS:	Material Intensity Per unit of Service
NIMBY:	Not-In-My-BackYard
SFA:	Substance Flow Analysis
UNCED:	United Nations Conference on Environment and Development
UNEP:	United Nations Environment Programme
WCED:	World Commission on Environment and Development

1. Research context

This thesis proposes a model for the assessment of sustainability of different waste management systems. Choosing the best way to deal with waste is a key issue if we are to attain a sustainable society. In most countries today, local waste management planning is the responsibility of the local municipal council, as is the implementation of Agenda 21 which was adopted by the nations of the world in Rio de Janeiro in 1992 (United Nations Conference on Environment and Development, 1992a). The first chapter of this thesis gives a background to the issue of waste, presents the concepts of sustainability and sustainable development and discusses municipal responsibility for local sustainable development.

The thesis consists of four parts. This first part introduces the context and relevant concepts and background to the study. The major research questions are presented. The second part gives a presentation of the methods used in the studies, and the theories behind them. Part three contains brief summaries of the empirical work described in more detail in the appended papers (I-IV). The fourth chapter analyses the results from the empirical applications, and presents the major conclusions. The chapter ends with an overview of the need for further research.

1.1 Waste generation and management

1.1.1 What is waste?

There are several definitions of waste. One with universal applicability could be that waste is the unwanted by-products of human activities. Such a definition is very comprehensive, and includes the solid, liquid and gaseous emissions from our societies. From a juridical point of view, however, the countries in the European Union define waste as follows:

> **"Waste" shall mean any substance or object in the categories set out in Annex I which the holder discards or intends or is required to discard.**

EU Council Directive 75/442/EEC on waste, as amended by Council Directive 91/156/EEC, Art.1(a)

In addition to this definition, there is a specific list naming 16 categories of waste. The last category, however, diminishes the value of the definition, since it includes “any materials, substances or products which are not contained in the above categories”! To reduce the uncertainties caused by this definition, a more detailed list, formerly known as the European Waste Catalogue (EWC), has been drawn up (European Commission, 2000).

In this list, each waste category is given a six-figure code for identification. The list is organised according to the sector in which the waste has its origin. This means that the same material, for instance wood, can have a variety of waste codes, depending on in which sector the waste has been generated. The advantage, of course, is that those involved in a specific industry only need to be familiar with the codes used within their own field. One important difference, compared to the general definition previously mentioned, is that the EU-definition does not define gaseous emissions as waste.

Even though the list is meant to be comprehensive, the definition of waste and the term ‘discard’ have been tested and more specifically defined by the European Court of Justice. One thing that has been clarified in court is that the commercial value of a substance, product or material, or the possibility of recycling it, does not influence its status as waste (Case C-359/88 [1990] ECR I-1509) in the eyes of the law.

1.1.2 A brief history of waste

Waste generation is not a new phenomenon. Among the most important sources of information for archaeologists studying ancient cultures and the lives of our oldest ancestors are their refuse dumps (Stewart *et al.*, 1994; McCorriston *et al.*, 2002). Through them we can learn about the tools they used, which materials were most important and their most common sources of food. Early industrial production can also be studied through the waste products they generated, even if most other traces of the production have been erased (Gordon, 1997; Hudson-Edwards *et al.*, 1999). As humans started to abandon their life as hunters and gatherers and started to cultivate crops, larger and larger populations were made possible. It was no longer necessary for everyone to work with the production or collection of food, and cultures and cities began to flourish.

When large numbers of people started to live in limited geographical areas, waste started to cause problems. During the Middle Ages, food refuse in the cities was one important factor responsible for the rapid spreading of the plague, since it attracted rats that hosted fleas that transmitted the infection to humans (McGovern *et al.*, 1997). As populations grew, all major cities needed to have some form of sanitation, not only to avoid diseases, but also to ensure that the nutrients in faeces and food refuse were returned to the productive soils outside the cities. This was the origin of waste management.

The materials used from the Middle Ages and up until modern times, such as leather, wood, wool, flax and hemp, were to a large extent natural and renewable. Products made of such materials were used and re-used until they could no longer be repaired, and the waste arising from this consumption could not reach sufficient volumes to cause any significant problems. There is also evidence that masonry materials were reused in new constructions, often more than once (Borradaile *et al.*, 1997). The metals that were used for anything other than coins and jewellery, primarily iron and copper, were so valuable that scrap was almost always taken care of and recycled into new products, if the old ones could not be repaired.

1930	1960	1990
• Bicycle	• Cross-country skis	• Alpine skis
• Housing (15 m ^{2})	• Camera	• Cross-country skis
• Food	• TV	• Computer
	• Telephone	• Camera
	• Bicycle	• TV
	• Fridge	• VCR
	• Electric stove	• Mobile phone
	• Washing machine	• Telephone
	• Toaster	• Mountain bike
	• Housing (28 m ^{2})	• Fridge
	• Food	• Dishwasher
		• Stereo
		• Electric stove
		• Washing machine
		• Summer cottage
		• Toaster
		• Housing (47 m^2)
		• Food (exotic fruits)

Figure 1. The growth of consumption. Examples of goods found in the average Swedish household, in three different years during the 20th century. (Kretsloppsdelegationen, 1997)

When industrialisation began on a large scale in the late 18th century, the negative side of increased material intensity became obvious. The transformation of materials into products requires energy, and lots of it

(Monkhouse, 2001). The use of unclean fossil fuels in combustion processes led to extensive pollution, and had considerable negative health effects. As the 20^{th} century progressed, a growing number of artificial, man-made materials were used, and products became increasingly short-lived in the consumer society. Landfill became the preferred solution to the waste problem (Hickman *et al.*, 1999). On many occasions it was combined with open furnace incineration of combustible wastes, to reduce the amounts that needed to be deposited as landfill. As environmental awareness grew in the 1960s, much of this combustion was forbidden, and the amount of landfill rose. In the 1970s it became more and more difficult to find suitable sites for new landfill. The Not-In-My-Backyard, or NIMBY, attitude has made it increasingly difficult to establish new landfill sites, while consumption patterns have continued the trend towards an intensified generation of waste (Figure 1).

1.1.3 Waste generation

The European Environment Agency has studied national waste statistics from a number of European countries, and calculated that the annual generation of waste in Europe is more than 3,000 million tonnes. Divided into regions, this is the equivalent of approximately 3.8 tonnes per capita in Western Europe, 4.4 tonnes per capita in Eastern Europe and 6.3 tonnes per capita in the most eastern Europe, the Caucasus and central Asian countries (European Environment Agency, 2003).

In the western Europe, construction and demolition waste accounts for 31 % of the waste generated, mining and quarrying waste 24%, industrial waste 15%, municipal waste 15% and waste from energy production 4% (European Environment Agency, 2003).

Waste treatment also contributes to a number of environmental problems, apart from the obvious resource depletion resulting from the waste generation.

- Collection and transportation of waste from households and firms to waste treatment facilities generate emissions contributing to global warming, eutrophication, acidification and a variety of emissions potentially damaging to human health. These aspects of waste management are important to consider, especially in sparsely populated areas, where collection routes will be long and treatment facilities distant.

- Landfilling of organic wastes will generate methane, which if not collected and incinerated is a powerful greenhouse gas. Landfilling of poorly source-separated wastes, or in poorly constructed landfills may lead to emissions of toxic substances through leakage. Landfills also require space, which is currently a limited resource in many countries, and threaten remaining non-exploited areas.
- Incineration of waste will contribute to global warming, since much of the generated waste is plastics from fossil oil. Incineration may also cause formation of dioxins, that are extremely hazardous to both humans and ecosystems, and may be released via flue-gases or ashes.

1.1.4 Modern waste management

During the past decades, much has happened to the way in which we regard and manage the waste produced in our society. Waste is something that concerns all of us, to a greater or lesser extent, and the treatment of waste will always have environmental, economic and social implications. Waste management has become an important scientific field, with journals of its own and with a wide range of international conferences focusing on everything from the collection of recyclable material to new wastewater treatment technologies.

Newly developed systems analysis methods such as Life Cycle Assessment, and Material Flow Accounting (these concepts and methods are further described in Chapter 2) have become increasingly important for our understanding of the pressure on the natural environment from our extraction, use and final disposal of resources. In combination with the growing concern for environmental protection, and deeper understanding of resource depletion issues, recycling of household wastes on a larger scale started to develop in the seventies. Paper and glass were among the first materials to be sorted and recycled into new products. At about the same time, efforts were made throughout Europe, to separate environmentally hazardous materials, such as batteries, mercury thermometers, medicines, chemicals, solvents and so on, from household waste.

In the late 80s, the polluter-pays-principle began to be enforced within the waste management field. Extended Producer Responsibility (EPR) was first introduced as a legal-term in France in 1975 (ADEME, 2003). In the early 90s the concept was used to target packaging materials. This was not because the packaging materials as such gave rise to environmental pollution, but mainly because they represented a large amount (both in volume and weight) of the municipal household waste, and would otherwise need to be deposited as landfill.

Germany was the first country to set up national targets for the re-use and recycling of packaging materials, which resulted in the Duales System Deutschland AG, and the famous Green Dot (Figure 2) which were both introduced in 1991. Although the Duales System was not a complete EPR-system, it influenced the first European Directive on packaging material, which was presented by the European Commission in 1992, and thereby the development of EPR-systems in many countries. All packaging was marked with symbols indicating how it should be sorted, and recycling bins for different fractions were distributed (Der Grüne Punkt, 2004). The system still relies heavily on central treatment facilities, where the waste is further sorted and controlled before recycling.

Figure 2. The Green Dot

In Sweden and Norway, EPR for packaging was introduced in the mid-90s as the national implementation of the European Packaging Directive from 1994 (Directive 94/62/EG). In Sweden this took place through legislation, and in Norway through agreements between the government and the packaging industry. In both these Nordic countries, the approach to recycling has been to facilitate extended source separation, rather than central sorting. Many fractions, such as paper and metal packaging, are also sorted after collection to enable recycling. In the case of paper there is also a strong economic incentive for this sorting, since homogenous fractions fetch a much higher price on the recycled paper market.

1.2 The concepts Sustainability and Sustainable Development

1.2.1 Sustainability

Even though the term hasn't been used until fairly recently, 'sustainability' has been a primary objective for communities throughout history. Nevertheless it has only been a key concept on the international political arena for 30 years. Different attempts have been made to define what a sustainable society is. Sustainability is sometimes confused with 'steady-state', stable or even static systems. Such systems may be sustainable as long as they are undisturbed, but sustainable systems are by no means the same as stable or static systems. An important concept here is resilience. Resilience is the ability to recover from, or adjust to disturbances. A resilient and sustainable system can be compared to a marble at the bottom of a bowl shaped as a semi-sphere. If something knocks the marble out of its position, it will after a while, resume its original position. The system is sustainable and resilient, and will return to the same point of equilibrium, given sufficient time.

Using the same example, if the bowl was turned upside-down, a marble could, with some effort, be made to rest on top of the bowl. This system is also in equilibrium, but the slightest disturbance will cause the marble to roll off. The marble will not return to its original position by itself. Such a system has a very low degree of resilience.

According to a dictionary, sustainable means the ability to be sustained. The question then is what a society needs to sustain to be sustainable. In relation to harvesting or using a resource, this is done in a sustainable manner if the resource itself 'is not depleted or permanently damaged' (Merriam Webster OnLine Dictionary, 2004). This resource-focused definition can be said to be the basis of the international efforts to promote sustainability. In the declaration from the 1972 UN-Conference in Stockholm (United Nations Conference on the Human Environment, 1972), the international community recognised the link between development and the environment, and the need to achieve economic and social development without depleting natural resources or damaging eco-systems.

Sustainability clearly has other dimensions than purely resource-related issues. The Natural Step has formulated four system conditions that describe a sustainable society (Holmberg *et al.*, 1996; Robèrt *et al.*, 2002). Their definition of sustainability is as follows (Robèrt *et al.*, 2002):

> In the sustainable society, nature is not subject to systematically increasing…
> 1. concentrations of substances from the earth's crust
> 2. concentrations of substances produced by society
> 3. degradation by physical means
>
> and, in that society
> 4. human needs are met world-wide.

1.2.2 Sustainable development

By definition, sustainable development is development that leads towards a sustainable society. The term 'sustainable development' became well known through the report from the Brundtland Commission, *Our Common Future* (WCED, 1987), in which it was defined as:

> "…development that meets the needs of the present without compromising the ability of future generations to meet their own needs"

This definition is often quoted, but it is also the topic of an ongoing debate. It has been criticised for being too anthropocentric and vague (Carter, 2001). The mere definition itself does not indicate the types of needs that must be fulfilled, nor does it include other needs than those of humans. Research to determine how to define the "needs" that must be fulfilled has been carried out, and reference is often made to the basic human needs as defined by Manfred Max-Neef and Abraham Maslow, e. g. Håland, (1999) and Vikman (2001).

Five years later, at the United Nations Conference on Environment and Development, *The Earth Summit*, in Rio de Janeiro, the nations of the world continued to work on the foundations laid in Stockholm 20 years earlier and again acknowledged the links between the environment and development. Once again, the need to ensure that development is obtained while preserving resources, eco-systems and biological diversity was stressed. In *Agenda 21* (United Nations Conference on Environment and Development, 1992a), which evolved from the Rio Conference, some principles are given that should be followed to ensure sustainable development. The four major sustainable development principles that emerged from the Earth Summit are the following (Mitchell *et al.*, 1995):

- **Environment**, to protect the integrity of eco-systems. According to this principle, nature and bio-diversity have a value in themselves, and should be preserved regardless of the "human needs" that we are aware of today.
- **Futurity**, (or intergenerational equity) to show concern for future generations and work to preserve natural and cultural capital. This principle is of course closely linked to the definition suggested by the Brundtland Commission.
- **Equity**, (intra-generational equity) to show concern for poor nations and disadvantaged populations. Closing the gap between rich and poor na-

tions, as well as between populations within individual nations, is essential for sustainable development. Unequal distribution of resources and wealth will lead to tensions that will prevent development.
- **Public participation**, allowing individuals to take part in decisions affecting them. Sustainable development cannot be enforced by undemocratic methods, but must be based on concerned individuals, who are given the opportunity to take part both in defining problems, and in suggesting possible sustainable solutions.

By committing themselves to *Agenda 21*, and by ratifying the Rio Declaration that was proclaimed simultaneously (United Nations Conference on Environment and Development, 1992b), the nations of the world took on responsibility for working with sustainable development issues from a bottom-up perspective, starting locally. Around the world, municipalities were given the role of co-ordinating activities at the local level and developing local Agenda 21 plans.

Ten years after Rio, the United Nations organised a new World Summit in Johannesburg, South Africa. Progress in some areas could be noted, but on the whole all the threats and obstacles to sustainable development were still present, and yet another political declaration was made (United Nations Environment Programme, 2002a). The nations signing the declaration also committed themselves to an implementation plan (United Nations Environment Programme, 2002b), further outlining measures that should be taken to eradicate poverty, to change unsustainable consumption and production patterns and to protect and manage our natural resource base.

1.2.3 Unsustainable patterns and the insufficiency of efficiency.

The recognition of the significance of unsustainable patterns of production and consumption in the Johannesburg Declaration, also expresses an understanding that technological development alone cannot solve the problem of the increasing impacts on the environment. More than 30 years ago, Ehrlich and Holdren (1971) presented their first equation describing the general relationship between consumption and environmental impact. Their formula has been further developed by over the years, and one useful interpretation is given by Azar et al (2002):

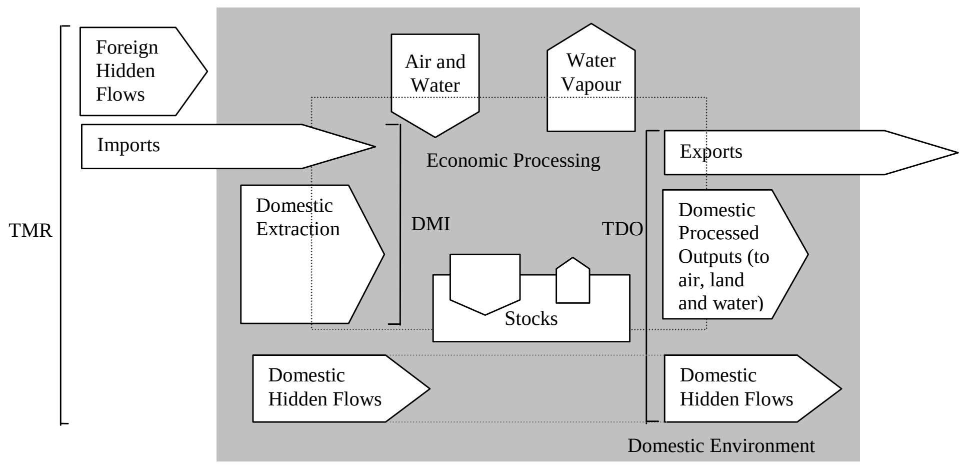
$$\text{Total impact} = i\ [\text{impact/kg}] * m\ [\text{kg/utility}] * u\ [\text{utility/capita}] * P\ [\text{capita}]$$

Ehrlich and Holdren had their primary focus on population growth and possible strategies to mitigate the environmental impacts arising from that. Even though population growth is still a major concern, the focus has shifted somewhat over the last decades, to what can be done to influence the other factors in the equation and thus achieve a ‘decoupling’ between economic growth and increased environmental impact (Azar et al, 2002).

The “m”, in the equation above, has given rise to concepts such as eco-efficiency, life cycle assessment, material intensity per unit of service (MIPS) etc. Indeed, western technology, research and development have largely concentrated on improving efficiency, and remarkable advances have been made in many fields. But there are a number of studies indicating that improved efficiency does not guarantee sustainable development or even reduced environmental impacts. One example is the Swedish Government Official Report on the efficient management of natural resources (SOU 2001:2, 2001). In their final report, the committee concluded that even though raw material consumption per unit produced in Sweden has decreased significantly (approximately 50%) since the middle of the 1950s, the growth in the volume of production has led to a 50% increase in raw material consumption from 1957 to 1996. In other words, the decrease in “m” has not been as rapid as the increase in “u”, in the equation.

Within the European Community, a set of national core indicators is used for the assessment of annual trends in waste management. The indicators provided by the European Topic Centre on Waste (ETC/W) have also been used in a report with a wider focus, which covers household consumption patterns as well as transport, energy, agriculture, waste and other topics (European Environment Agency, 2001). They have shown that there is a statistically significant correlation between economic activity measured in Euro per capita, and the generation of municipal and construction waste (Munck-Kampmann, 2001). The ETC/W also concludes that in some countries the reduction in waste generation per unit of production through technological improvements is overshadowed by a growth in the total quantity of goods produced and consumed (Munck-Kampmann, 2001).

This conclusion is also supported by a comprehensive report published by the World Resource Institute on material flows (Matthews *et al.*, 2000). Their report states that no evidence could be found of an absolute reduction in resource throughput, even if there were signs of a decoupling between economic growth per capita and resource throughput.



TMR (Total Material Requirement)= DMI+Domestic Hidden Flows + Foreign Hidden Flows
DMI (Direct Material Input)= Domestic Extraction + Imports
TDO (Total Domestic Output)=DPO+Domestic Hidden Flows
NAS (Net Additions to Stock)= DMI –DPO – Exports
DPO (Domestic Processed Output)=DMI-Net Additions to Stock -Exports

Figure 3. The Material Circle. System boundaries and indicators in the World Resources Institute material flow analysis of industrialised economies (Matthews *et al.*, 2000). The grey area symbolises country boarders, and the dotted square the economic society.

In their study of the economies of Japan, Germany, Austria, the Netherlands and the United States, they found that one half to three quarters of the annual input of resources in industrial economies is returned to the environment as waste within a year. The study used a classical material flow accounting approach, which has been suggested as an appropriate tool to use for decision making about industrial ecology (Kleijn *et al.*, 2001). Examples of the indicators used are shown in Figure 3 (Matthews *et al.*, 2000).

In conclusion, there are a number of indications that improved efficiency have not resulted in decreased environmental impact, but rather in increased production volumes and levels of consumption. These are unsustainable patterns of resource use that need to be addressed.

1.2.4 Planning for sustainable waste management

In *Agenda 21*, which was adopted in Rio (United Nations Conference on Environment and Development, 1992a), Chapter 21 deals with environmentally sound management of solid waste and sewage. This is not surprising, since how we manage our waste is a determining factor for our ability to attain an environmentally sustainable society (Figure 4).

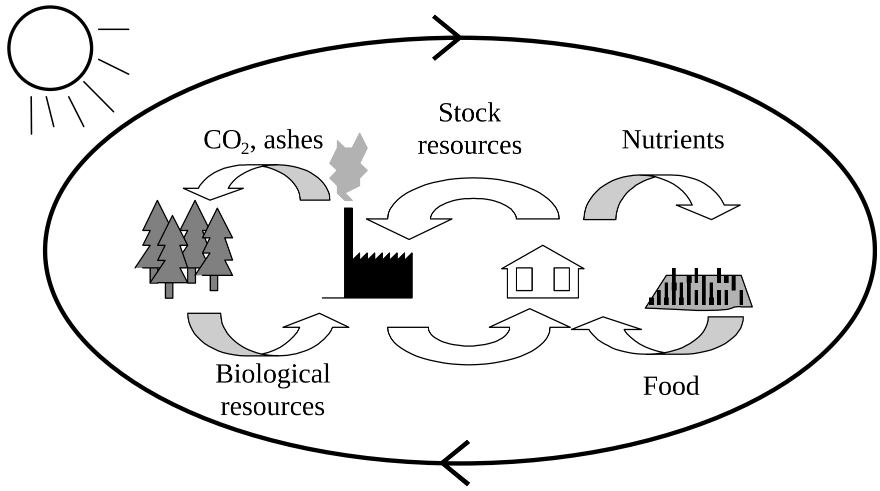


Figure 4. Loops that need to be completed in order to achieve an ecologically sustainable society (adapted from Tiberg (1993))

One way to interpret the suggested loops of Figure 4 is:

- Emissions of 'molecular wastes' such as carbon dioxide must be in balance with the natural uptake and sequestration of these substances in order not to disturb the Earth's climate system.
- Non-renewable substances should be confined to closed material loops within our society to preserve them for future generations, and to protect natural ecosystems.
- Nutrients in food waste and wastewater sludge must be returned to productive agricultural soils (not necessarily in direct loops, however).

The four system conditions developed by The Natural Step (Holmberg *et al.*, 1996; Robèrt *et al.*, 2002), cited in paragraph 1.2.1, emphasise again the loops in Figure 4. Conditions 1 and 2 are directly related to waste management and the necessity of establishing material cycles, in accordance with the principles of Industrial Ecology (Brattebø *et al.*, 1999). Condition 3 also has bearing on waste management, since waste treatment facilities must be located somewhere, and natural eco-systems might be disturbed.

In Sweden, municipalities have been made legally responsible for the promotion of acceptable and equal living conditions for all citizens and for securing a long-term sustainable environment for present and future generations (Code of Planning and Construction, SFS 1987:10).

Municipal planning can be divided into physical and social planning, and between them one could place technical planning. The above-mentioned law, however, focuses on physical planning and construction, and on the use and protection of natural resources. Social planning, on the other hand, deals with almost all the other aspects of municipal responsibility, such as social welfare, childcare, education, care of the elderly etc. Technical plans, such as energy plans and waste management plans, are closely linked to both physical and social planning.

Waste management planning is also considered to be the responsibility of municipal authorities. In Sweden all municipalities are obliged by the Environmental Code (SFS 1998:643) to formulate and maintain a management plan for all waste that arises within their geographical borders, not only household waste for which the municipality must also provide a collection service.

Waste management is a service provided for the local inhabitants, private and public companies and institutions, so it must be formulated in interaction with the community and must consider social aspects, such as the level of service, the accessibility to source-separation for disabled citizens etc. It is also evident that social acceptance and willingness to participate in recycling activities is crucial for the success of any waste management programme (Klang, 2003) so a waste management plan must also contain strategies for communication and dissemination of information. At the same time waste management requires physical areas for waste treatment, composting plants, sewage sludge treatment, landfills, incineration plants etc. Such facilities must be located so that they do not disturb residents. Waste management planning must therefore be a part of the physical planning process as well.

According to the Environmental Code, waste management plans must contain descriptions of measures that the municipality will undertake to reduce the amount and level of harmfulness of the waste. Plans should be updated regularly, and be submitted to the county administration board. Further instructions about waste management plans are provided by the Swedish Environmental Protection Agency. These instructions (NFS 1999:6) specify that the waste management plan should divide the waste into the following categories, and indicate the amount, collection methods and treatment methods for each category:

1. Municipal solid waste
2. Park and garden waste
3. Construction and demolition waste
4. Waste from energy production (slag and ashes etc.)
5. Municipal sewage sludge
6. Industrial sewage sludge
7. Mining and mineral extraction waste
8. Branch-specific industrial waste
9. Non-branch-specific industrial waste
10. Special waste (including hazardous waste)

1.3 Structure of this study

1.3.1 Scope of study and objectives

Sustainability and sustainable development are multi-facetted concepts, and applying them to waste management provides some interesting challenges. The purpose of this thesis has been to investigate how a tool to assess different waste management systems should be developed, to be valid, reliable, useful and efficient for waste management planners, and at the same time contribute to sustainable development. The tool should also ensure that no important aspects of sustainability, as we understand this concept today, are omitted or overseen. The objective has been to improve methodologies that can support decision-making processes in sustainable waste management.

1.3.2 Research questions and delimitations

This research has focused on two major questions. The first is directly related to the subject mentioned above:

- Which aspects of waste management are most important to include in a sustainability assessment, according to the practitioners?

- What are the essential properties of a sustainability assessment model for municipal waste management, to ensure that all aspects of sustainability are considered?

Focus has been set on the special conditions of rural and sparsely populated areas in northern Scandinavia, and on population centres and municipalities within this area.

1.3.3 Sustainability versus sustainable development assessment

The point of departure for this study has been that it should be possible to develop a sustainability assessment tool for waste management. Sustainability assessment can be done in two different ways, which can be defined as relative or absolute. An absolute assessment examines a certain waste management system to see if it meets certain criteria which define sustainability (for instance the four system conditions mentioned above), and the results of the assessment are of a binary character: ‘sustainable’ or ‘not sustainable’.

A relative assessment compares one or more systems with each other, as well as measuring them against chosen sustainability criteria. The system that least violates the criteria for sustainability can then, under certain conditions, be assumed to have a better potential to contribute to sustainable development. This has been described by (Robèrt, 2000), as the sustainable opening of a narrowing funnel.

The assumption that an alternative that least violates the sustainability criteria will be more likely to contribute to sustainable development is, of course, not always correct. One system can, for example, use less fossil fuel than another, but not have any potential to develop over time and be converted to run on biofuel instead, and this aspect must be considered when different options are compared.

Determining when a sustainable state is actually reached can be difficult. Absolute sustainability is only easily analysed in aftermath, when the impacts of a system has been studied for some time. The objective of this thesis work is therefore to develop a tool to assess different options ability to contribute to a sustainable development of the waste management system.

Sustainable development in waste management is understood to be a development that brings the waste management system closer to a sustainable system (a system completely fulfilling all systems conditions), than the current system is. The end goal is to reach a sustainable waste management, but since the exact point of sustainability is unknown to us, the assessment tool is instead focused on the development of the system.

2. Theory and methods

2.1 The fundamentals of systems theory and systems analysis

A system can be viewed as a number of components united in a totality. The components and their relationships are distinguished from the rest of the world (the environment) by the system boundaries. The study of the function of systems is called systems theory. A Systems Analysis (SA) is a special form of scientific, engineering problem-solving technique, focused not on investigating or describing the individual components in detail, but on the interconnections between the components and on the totality itself (Gustavsson *et al.*, 1982).

In order to make a systems analysis, it is necessary to define the system boundaries, showing which components belong to the system studied, and which do not. All technical systems are influenced by factors outside the system boundaries, and also influence their environment. This interaction between the system and its environment is described by input and output parameters Figure 5. One single system may have many different kinds of system boundaries, depending on the complexity of the systems analysis. Some examples are physical, temporal, social and economic boundaries.

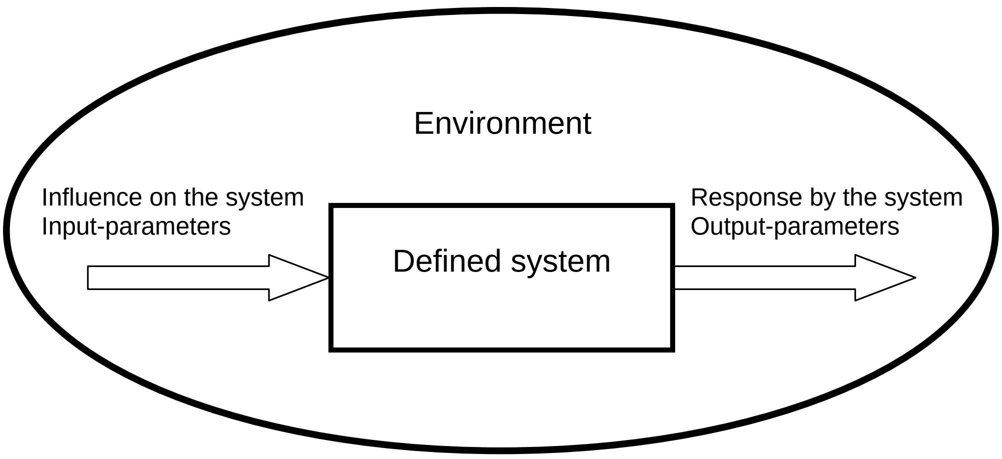


Figure 5. System interaction with environment. Adapted from (Gustavsson *et al.*, 1982).

In this way, a model is constructed. A model is a simplified representation of the real system, omitting components that are of small importance for the system response (output parameters). The input- and output parameters described in Figure 5 can sometimes be in the form of a material flow (e.g. an input of fuel to a combustion engine and an output of various emissions), but the arrows in the figure can also illustrate any other type of system-environment interaction. A prerequisite for performing an accurate systems analysis, is that the components within the system are sufficiently well known and described, so that the system response can be accurately estimated. For a waste systems analysis, components could, for instance, be different treatment methods, collection vehicle performance, factors influencing source separation efficiency in households and so on.

2.2 Tools to assess and monitor sustainable development

If we accept the definition of a sustainable society given by The Natural Step (Holmberg *et al.*, 1996), referred to in Chapter 1.2.1, we still need instruments to monitor to what extent different activities contribute to, or obstruct sustainable development. Several tools have been developed during the past decades, some of which could be used for this purpose.

One can categorise tools as being either analytical or procedural (Wrisberg *et al.*, 2002). Procedural tools are focused on guiding the process to reach and implement environmental decisions, while analytical tools model systems to provide technical information for decision making (Wrisberg *et al.*, 2002). Some tools that have been used in different waste studies will be described briefly here, together with short comments regarding how the methods have been used in the development of the assessment model.

2.2.1 Analytical tools

Life Cycle Assessments (LCA)
A Life Cycle Assessment aims to describe the environmental impact of a product or service, from ‘cradle to grave’. The objective is to use this information to improve efficiency and reduce environmental impacts in the entire life-cycle chain, and thereby avoid sub-optimisation. Life Cycle Assessments have often been used in waste management studies (Barton *et al.*, 1996; Finnveden, 1999; Sundqvist, 1999; Clift *et al.*, 2000) There are international standards for how to perform an LCA (ISO 14040 – 14043),

and this methodology is frequently used in research and industry (Andersson *et al.*, 1998; Jönsson *et al.*, 1998; Erlandsson *et al.*, 2003). An LCA, however, focuses primarily on describing environmental aspects (sometimes including human-health aspects), and does not address other aspects of sustainability and sustainable development.

Application in the suggested assessment model

The treatment of waste will influence production and extraction of virgin resources, and it was therefore clear from the beginning of the development process for the assessment model, that environmental aspects must be described using a life cycle perspective. LCA studies and data have been used to calculate all environmental aspects in the developed model.

Material Flow Accounting (MFA)

MFA can be carried out on many different scales. As shown in Chapter 1.2.3 it can be used to describe the complete material metabolism of entire nations (Matthews *et al.*, 2000; Daniels, 2002). Such studies are often referred to as bulk-MFA (Wrisberg *et al.*, 2002). It has also been used successfully to determine the final fate of hazardous compounds such as heavy metals and pesticides (Kleijn *et al.*, 2001). This specific type of MFA can also be called Substance Flow Analysis, or SFA (Wrisberg *et al.*, 2002). An MFA does not consider the costs, and seldom the environmental or social effects of the material flows studied, even though it is very useful for identifying substances that may cause acute problems in the future. The LCA-method does to some extent contain MFA, with the difference that an LCA often won't consider to which specific ecosystems emissions are released, and therefore report potential, rather than actual environmental impacts. An LCA, will also typically, be more focused on yearly emissions, while the MFA always also consider the formation of stocks of different substances in the system, that may be released on a later stage.

Application in the suggested assessment model

MFA has not been used in the developed assessment model. It could provide additional data of interest, by improving the knowledge about the performance of different system components. One example would be to analyse temporal and spatial differences in release of heavy metals to the environment, depending on choice of treatment method (discussed in Paper II).

Emergy Analysis

Emergy Analysis has been developed by Howard T. Odum and co-workers, during the last three decades (Odum, 1971; (Odum, 1996; Odum *et al.*, 2000). In emergy accounting the energy content and mass of all inputs to the system are calculated. These actual inputs are then multiplied by factors – called transformities – which is a measure of the solar energy needed to support one actual unit in the system. The unit achieved through this operation is solar emergy joules, or *sej*. The method has been developed to include monetary flows, which are converted to Emergydollars (Em$). Resources that must be purchased and transported will have a much higher emergy content than free-flowing local resources such as sunlight and rain. Emergy sustainability indexes can be calculated to assess and compare the relative sustainability of alternative technological solutions (Paper III). The major advantage of the method is that it is developed as a tool for systems analysis, and it uses a life-cycle perspective to calculate the transformities by which the physical energy content is transformed to emergy. The drawback of the method is that it is not easily understood by persons unfamiliar with the emergy-concept, and the results are on such a high level of aggregation that they may be difficult to use for decision-makers.

Application in the suggested assessment model

Emergy analysis was one of two methods used to assess the relative sustainability of different wastewater treatment technologies described in Paper III. The conclusion is that the method is not sufficiently transparent and for this reason not completely appropriate in its current form, to use in municipal decision-making where the basis for the decision must also be communicated to the public.

Socio-ecological indicators

Some of the originators of the socio-ecological principles (Chapter 1.2.1) have also suggested a set of indicators, which have been developed to assess to what extent each of the four System Conditions is violated (Azar *et al.*, 1996). They can be used to assess the performance of nations, or to compare different technological options to one another (tested in Paper III). However, the indicators suggested do not automatically embrace a life-cycle perspective, which may cause the environmental effects from the use of certain elements and substances to be underestimated, if the indicators are used without sufficient insight into the entire life cycle of the material. Their major advantage is that they are closely linked to a theoretical and pedagogic definition of sustainability.

Application in the suggested assessment model
Socio-ecological indicators were also used in the study described in Paper III. Although the indicators are linked to a widely spread and easily understandable definition of sustainability, and embrace a holistic view of sustainability, it is difficult to assess many of the specific aspects expressed as important by practitioners for evaluation of waste management options (see Paper IV). Economy and work environment are two examples of such aspects.

Other analytical tools
The tools mentioned here were chosen since they have been used in waste management systems analysis, and proven to provide important results. There are other analytical tools that could be used, such as for instance Environmental Risk Assessment (ERA) and Cumulative Energy Requirements Analysis. For closer descriptions of these and other tools, please refer to Wrisberg *et al.* (2002).

2.2.2 Procedural tools

PICABUE
The PICABUE-system is a framework for the process of developing sustainability indicators in co-operation with stakeholders (Mitchell *et al.*, 1995). The name of the framework is actually an abbreviation consisting of the first letter of the keywords describing each step that is carried out in the indicator-formulating process. The seven steps are:

- Stakeholders to reach consensus on: **P**rinciples of sustainable development, Objectives of indicator use.
- Identify and select **I**ssues of concern.
- **C**onstruct / select base indicators of quality-of-life issues of concern
- **A**ugment quality-of-life indicators with reference to sustainability principles to produce sustainability indicators.
- Modify sustainability indicators to account for **B**oundary difficulties.
- Supplement sustainability indicators with **U**ncertainty indicators.
- **E**valuate final sustainability indicators with respect to: Desired indicator characteristics; Objectives of indicator programme.

The indicators resulting from this work, focused on quality-of-life (meaning human life) will inevitably be very anthropocentric. To compensate for this, a parallel process to develop complementary ecological indicators, acknowledging the intrinsic values of the eco-systems, must be carried out (by experts). The strength of the PICABUE-method lies foremost in the democratic process hopefully leading to a consensus among stakeholders on what the issues of concern should be. This is, of course, also a difficulty with the system, since stakeholders often have conflicting opinions regarding which issues should be viewed as important.

Application in the suggested assessment model

The PICABUE-system has not been used to develop indicators in the assessment model. It was considered to be important to assure that each stakeholder has the freedom to draw his or her own conclusions, based on personal value sets and on the indicators he or she regard as the important ones. The indicators have instead been chosen after dialogue with stakeholders, and through questionnaire investigations. This doesn’t necessarily result in stakeholder consensus on what are the issues of concern.

Environmental management systems (EMAS and ISO 14001)

Environmental management systems (EMS) originate in the same management tradition as quality management, and there are many similarities between the international standards regulating these two systems, but the type of indicators used is not regulated by the standards. An EMS is based on an environmental investigation, where the most important environmental aspects are identified. Environmental objectives to reduce negative impacts are then set, and instruments of control to monitor and assess improvements are defined. The core idea with both environmental and quality management is to work with continuous improvements, but the standard does not indicate what type of tools or indicators should be used to monitor the improvements. Several municipalities use EMS to organise and monitor their environmental performance (Emilsson *et al.*, 2002).

Application in the suggested assessment model

Environmental management systems can be used by a local waste management operation, to set goals and monitor development for all activities, perhaps in combination with material accounting tools (Burström, 1999). It is possible to include objectives relating to both external and internal environmental issues, and in combination with economic monitoring instruments it can be very useful for systematic management. It is, however, a procedural tool dedicated to monitoring and management of existing operations, and constant improvements of those operations. It cannot be used to assess possible alternative options.

Other procedural tools
As with the analytical tools, these are not the only plausible procedural tools to use in waste management analysis. Environmental Impact Assessment (EIA), is one other example of a procedural tool that can be used, for instance when assessing the environmental impact of a new proposed waste treatment facility (Wilkins, 2003).

2.3 Sustainable Development Indicators

In *Agenda-21*, adopted in Rio de Janeiro in 1992 (United Nations Conference on Environment and Development, 1992a), Chapter 40 is concerned with ‘Information for decision-making’. In paragraph 40.4 it is stated that:

> ‘Indicators of sustainable development need to be developed to provide solid bases for decision-making at all levels and to contribute to a self-regulating sustainability of integrated environment and development systems.’

In the passages following this statement nations, governmental and non-governmental organisations are urged to participate in the development of indicators of sustainable development. The scientific community had already begun working on this, but as a result of the Rio Declaration the work was intensified in the early 90s.

Sustainability indicators can be been developed on many different levels of aggregation (Figure 6). Some indicators can be unprocessed data values, others can consist of a chosen selection of data of key importance (i.e. certain data may be excluded from further analysis, even if it is readily available or possible to obtain).

Composite indicators, consisting of several data recalculated to a common unit, can be used to improve comparability and understanding. An example of a composite indicator commonly used in LCA-studies is CO_2-equivalents that convert data on emissions of CO_2, CO, CH_4 and other greenhouse-gases to a single indicator. Some data values can be included in more than one composite indicator. For instance, emissions of NO_x can contribute to acidification, and be combined with emissions of SO_x and other acidifying substances in a composite indicator for potential acidification. But NO_x may also contribute to eutrophication, and be included in the composite indicator for potential eutrophication or maximum oxygen demand (Figure 6).

When indicators of different types are related to one another, ratio-indicators are obtained. Eco-efficiency, relating the economic value of a production process to the environmental impacts the production causes, is a well-known type of ratio indicator.

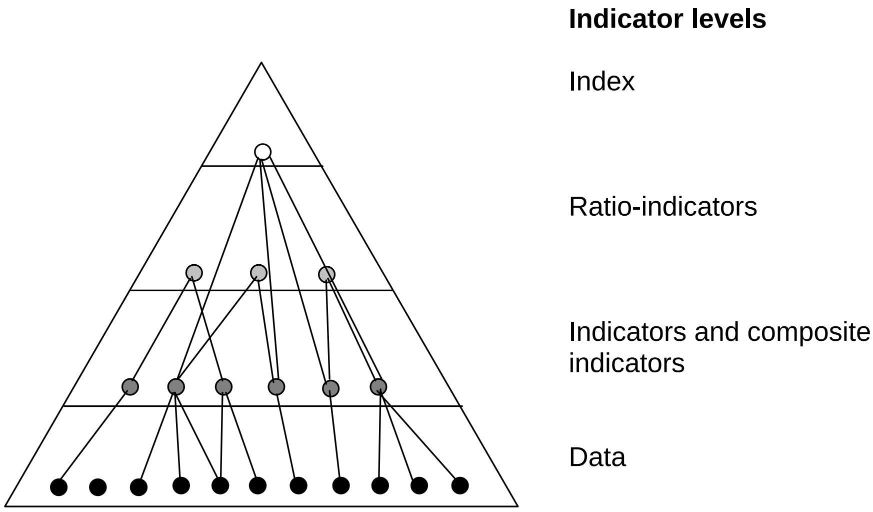


Figure 6. Different levels of aggregation of data in sustainability indicators

Some sustainability assessment methods suggest an even higher level of aggregation, and promote a single index describing all aspects of the system studied. This index can be based on all the selected indicators and composite indicators, or a chosen subset. Ecological footprints (Wackernagel *et al.*, 1996; Wackernagel *et al.*, 1998; Wackernagel, 2001), convert data on resource use and waste generation to the area of productive land needed to provide the resources and take care of the waste.

The major advantages of this method is that it easily understood by, and communicated to the general public and that it clearly demonstrates that since the surface of the earth is finite, there must be limits to our use of resources and our generation of waste, if we are to attain sustainable development (Wackernagel *et al.*, 1998). The method is also effective in illustrating the gap between rich and poor countries, and the immense injustices in the distribution of resource consumption around the globe (Wackernagel *et al.*, 1997).

Another single parameter index that is commonly used is the gross domestic product, GDP or GDP per capita, which is often used as an indicator of national wealth and well-being. When studied in closer detail, GDP really only indicates the speed by which resources are converted into money flows, and tells us very little about how this is done or how resource use is distributed among the population (Bossel, 1999).

The problem with both GDP and the ecological footprints, is that the level of aggregation is so high, that these indexes are not useful as the basis for decision making when action is called for on a project level.

In life-cycle assessment, the step where indicators of totally different aspects are recalculated to a single index is referred to as the weighting step. Several methods for performing weighting exist, e. g. the Eco-scarcity (Ahbe *et al.*, 1990) and EPS-methods (Steen *et al.*, 1992). Weighting is not a mandatory step according to the ISO-standard for life-cycle assessments (ISO 14040 – 14043), and in some cases it is even forbidden. The reason for this is that all weighting methods are heavily value-based. By choosing a method subjectively, the final outcome of the assessment can be unduly influenced, making it inappropriate for anything other than internal use (Lindahl *et al.*, 2002).

Responsible decision-making requires enough information, with sufficient transparency, to enable the decision-maker to reach his or her decision based on his or her personal set of values. This is particularly important in any kind of political or public policy process, since voters in a representative democracy should be able to expect elected representatives to think for themselves, and not let ‘experts’ control final decisions. In the development of the assessment framework for this study it was therefore decided that ratio-indicators would be the highest level of aggregation in this model. We also decided to include aspects related to all three areas of sustainability; environmental, economic and social aspects.

2.4 Theoretical model for waste management planning

Rules and regulations determine what a municipal waste management plan must contain, but they do not give detailed information about how the process of developing the plan should take place. It is important to have a basic theory for this in order to be able to develop tools that can be useful in facilitating and improving such processes. The following is a presentation of an ideal planning process, based on a general systems engineering process model.

The capital letters in parentheses in the following paragraphs all refer to Figure 7. There are a number of stakeholders (A) in all waste management planning situations. Stakeholders can include the general public, organisations and companies, and authorities with waste management responsibilities. Different stakeholders will express different demands (B) that they want the waste management system to fulfil. These demands can be clearly expressed as rules or legislation passed by the responsible authorities, or they can be expectations from households and firms, or explicit demands from refuse disposal personnel. The demands can be roughly divided into three groups: economic, environmental and social aspects. Some demands may very well be contradictory to others.

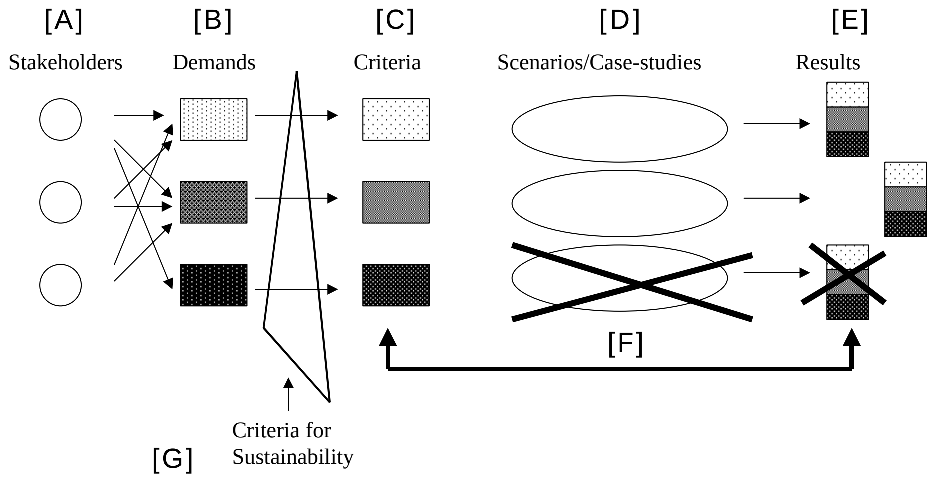


Figure 7. General systems engineering representation of a possible waste management planning process (modified from an original figure by Helge Brattebø).

It is now up to the waste management planning authority (typically a municipal council and administration) to translate these demands into criteria (C) that the waste management system must fulfil. These criteria are also divided into three groups, but since they will have to be decided through compromises and negotiations with the stakeholders (apart from demands specifically determined by law) they will not fully match all the demands expressed. The criteria must, however, at the very least be in accordance with the legislative demands from the authorities.

The time has then come to examine the possible options in terms of organisation, source separation, treatment methods and so forth. A decision must also be made about the period of time that the plan will be valid. When this is done, relevant scenarios can be formulated (D). An analysis of these scenarios can then be made. Such an analysis will typically make combined use of both qualitative and quantitative methods. If the analysis is a systems analysis leading to a sustainability assessment, the outcome will provide results (E) that can be compared (F) with the criteria formulated, based on the demands expressed by the stakeholders. Scenarios not fulfilling the criteria can be ruled out from further studies, leaving the decision makers with fewer alternatives to choose between.

The process from B to E and F may be a highly iterative one, since the formulation of demands and criteria will also depend on the possible results from realistic technical solutions and scenarios.

An objective of this thesis, and the empirical work that it is based on, has been to formulate criteria for sustainable waste management (G), and appropriate indicators to monitor sustainable development within waste management systems. The idea has been that such criteria should influence the formulation of systems criteria for the waste management system, but also to use sustainability indicators to compare different scenarios with each other, and with the boundary-criteria set up in the planning process. Finally, an optimal planning process is iterative, and might have to go several rounds, allowing the results of the analysis to influence the formulation of new criteria, until a final selection of alternatives can be made. Up until this point, municipal officers will have done most of the practical work, but politicians, in some form of municipality board, will take the final decision. As ultimately responsible for the decision, politicians will have a special role also as stakeholders, and on the final formulation of criteria.

In practice, however, waste management planning processes may not follow a scheme similar to the representation in Figure 7. Time and financial resources are often limited, making it difficult to obtain sufficient input from stakeholders in the process. This may cause an emphasis on clearly expressed, absolute demands, such as those expressed in legislation, and on financial aspects, since they often are relatively easy to assess. In reality, the choice of technology is often limited to systems that are well-known and tested, as this allows greater predictability about investment costs and the operational reliability.

2.5 Sustainability assessment in waste systems analysis

Traditionally, the sphere of waste management has been limited to the end stage of the life-cycles of different products. To assess the sustainability of a waste management system, it is necessary to widen the scope of waste management, and include its interaction with resource extraction, production, consumption and all emissions to the environment measured on a life-cycle basis. As a result of this widening, most of the systems studied will be outside the waste managers' sphere of control (Figure 8). They have very limited opportunities to influence the production and consumption processes, but even so, these components will be affected by the choices made in waste management, and must therefore be included in the model to make a correct assessment of the sustainability of the available options for waste management.

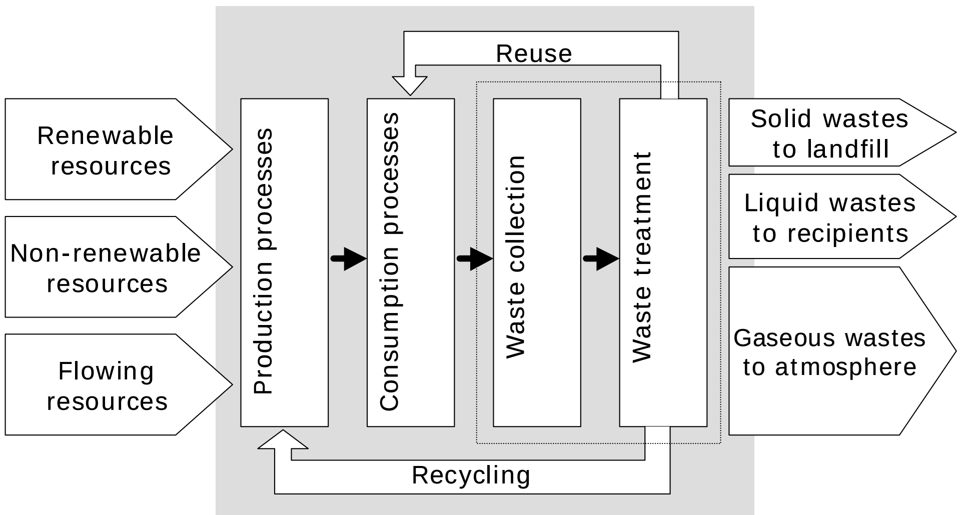


Figure 8. Expanding the scope of the waste management plan. The dotted line indicates the traditional scope of waste management plans, the grey area the technosphere. The arrows are not to scale.

Analysing a system such as the one described in Figure 8 can be done by comparing different scenarios. Some authors distinguish between 'internal' and 'external' scenarios, where the internal scenarios refer to different variations of the waste treatment systems, and the external scenarios refer to possible variations of the external parameters that are outside the waste managers' sphere of control (Eriksson *et al.*, 2003).

A crucial step in the definition of the system components, is to decide the design of the complementary or compensatory systems (Figure 9). In a waste systems analysis, the waste can be used to fulfil different functions. For instance, biological waste can be incinerated in a combined heat and power plant to produce heat and electricity. Or it can be taken to an anaerobic digestion plant that produces methane (which can replace diesel fuel), and a nutrient rich digestion sludge that can be used for soil improvement.

If the waste is landfilled instead, other methods must be used to produce heat, electricity and fuel, and the nutrients to improve soil quality must be obtained by other means. Often, the definition of the complementary system will decide the outcome of the analysis (Paper II). An important factor in this is the time aspect. It is very probable that the nature of the complementary systems will change over time, as present technology evolves and is improved and new technology is developed.

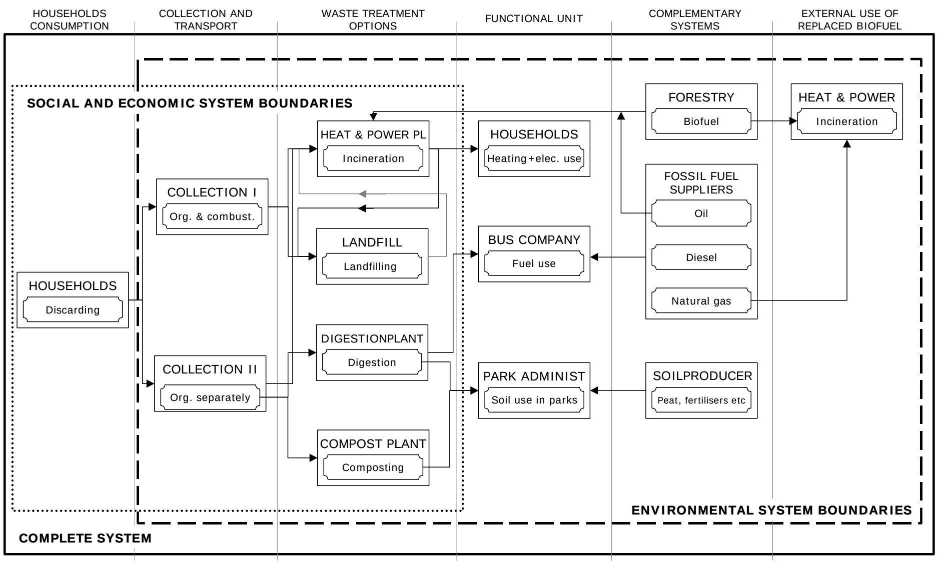


Figure 9. Example of system boundaries and complementary systems (from Paper II)

2.6 *Sustainable development indicators for waste management*

2.6.1 Conceptual framework

The complexity of an analysis will increase with the number of aspects that are considered. To ensure validity in a sustainability assessment, it is necessary to include environmental, economic and social parameters. For a further discussion on validity, see chapter 4.2. To maintain validity, and at the same time ensure the practical usability for practitioners, a framework model can be used to organise the aspects studied in a systematic manner.

For this study, the concept of the triple-bottom-line approach (Elkington, 1997; Rogers *et al.*, 2001), was used to construct a model framework of indicators of different types of aspects. The main aspects of sustainability suggested in this model are the environmental, economic and social aspects. Environmental and social aspects are clearly identified in *Our Common Future* (WCED, 1987) and *Agenda 21* (UNCED, 1992a), as important to consider to be able to obtain sustainable development. Economic aspects are important to include since it will have a great importance for municipality decision-making. For each of these main aspects sets of indicators were developed.

When indicators of different aspects are linked to one another, the result is a new type of aggregated ratio-indicator. When an environmental indicator is linked to an economic one, an eco-efficiency indicator is obtained. Economic indicators can be related to social indicators, to produce socio-economic indicators. Finally, when environmental indicators are related to social ones, indicators of environmental awareness can be obtained (Figure 10). For examples of indicators used, see chapter 4.4.

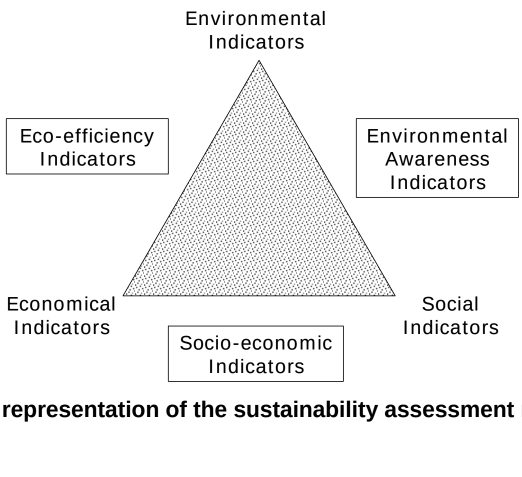


Figure 10. Graphic representation of the sustainability assessment model

2.6.2 Interpretation of results

When evaluating the results obtained for different indicators, it is important that the cornerstone aspects aren't considered to be interchangeable. This can be described as a three dimensional system of coordinates, where the three axes only meet at the origin (Figure 11).

If an alternative technology or waste management option is compared to an existing, the existing one can be said to be at the origin in the system of coordinates. The position of the alternative technology in the 3D-plot in Figure 11 will then demonstrate its relative qualities (relative sustainability) in relation to the reference technology, which is now situated in the origo of the diagram. If the alternative is positioned positively on all three axes, the assessment is that it is closer to sustainability and hence, such a re-positioning could be called a sustainable development. If the alternative is better positioned on only one or two axes, the result is ambiguous.

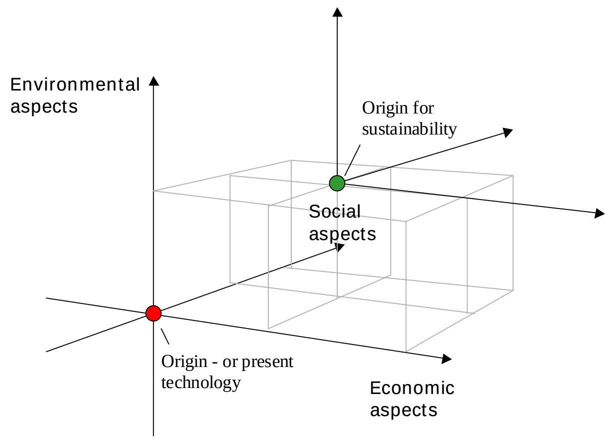


Figure 11. Sustainability and sustainable development. A development leading from the present origin positively along all three axes is sustainable.

In that case it is important to remember that environmental and social aspects have precedence over economic aspects from a sustainability point of view. This means that an option must be better or equally positioned on the environment and social axes, to be considered to contribute to sustainable development, and that it's position on the economy axes is of subordinate importance. Indicator values can also be used to analyse what can be done to improve the alternative's performance, at least up to the point of origin.

If no such measures can be thought of, it will be up to decision-makers to value the impact of different aspects, and form their decision based on this valuation.

This can for instance result in that a more expensive solution is regarded as more favourable than the existing one, because it is better from environmental and social aspects. The final decision may also be that an economically favourable option is chosen, regardless of social and environmental aspects. In that case, the indicators will serve to show the negative influence on sustainable development, which should be balanced with other measures to justify the decision. At some level, which is not at all easily defined, in the system of co-ordinates, sustainability is reached. Sustainability, however, is not a finite state, but can be viewed as a new origin in the system of co-ordinates, which we seek to reach. Once there, only options that maintain or improve environmental and social conditions can be regarded as sustainable.

3. Empirical work

3.1 Results from the case-studies

The papers included cover a series of case-studies which have been used to develop methods and a framework for a sustainability analysis of waste management methods. In this chapter, a brief summary of the results is given. The case studies were chosen to represent all three of the major waste fractions in the sparsely populated region of northern Scandinavia. Earlier studies have shown that the waste that arises in this region can be roughly divided into three fractions of similar total weights (Wadman *et al.*, 1997):

1. Construction and industrial waste
2. Solid household waste
3. Sludge from waste water treatment plants.

The three first case-studies were therefore devoted to one of these fractions each. A fourth study was conducted to investigate how decision-makers and municipal officers perceive a waste systems analysis, and how it functions as a decision support tool.

3.2 Sustainable management of demolition waste

The first case-study focused on the possibilities of developing a framework model to evaluate the sustainability of different ways of increasing the recovery of construction and demolition waste. Construction and demolition waste is one of the largest fractions of waste brought to landfills throughout the European Union today (Symonds Group Ltd, 1999). This case-study focused on a project which aimed to reintroduce groups of long-term unemployed persons to the labour market, by offering them vocational training in the form of environmental studies and practical work with the recovery and recycling of construction and demolition waste. One reason that construction and demolition waste is not recovered more often is that careful dismantling of constructions requires extra time, and time is a limiting factor in modern construction. When evaluating the sustainability of jobs created within this sector, it was therefore crucial to take eco-efficiency aspects into account. This was not done in the traditional sense, where eco-efficiency is defined as the product or service value, divided by the environmental impacts (OECD, 1998), but rather as the negative environmental impacts avoided, divided by the hours of labour invested. This form of eco-efficiency indicator made it possible to see where the limited resources of labour and time could be used to achieve the largest possible environmental benefits. Since time is often a limiting factor in construction and demolition work, such eco-efficiency indicators can be very useful as optimisation tools. Three different activities were studied in detail: cleansing mortar from bricks to enable re-use, recycling steel, and the re-use of sanitary porcelain.

To summarise the results, brick-cleansing turned out to be the most eco-efficient activity of the three, i.e. it led to the largest reduction in negative environmental impacts. However, brick-cleansing was not sustainable from a social perspective, since several of the steps in the cleansing process required manual handling of the heavy bricks, which in the long run could lead to musculo-skeletal disorders. As a result of the project, ways of improving the physical working conditions during brick-cleansing were suggested. The case-study also showed that steel-recycling was only barely sustainable from an economic perspective, whereas the recovery of steel products for re-use was definitively economically sustainable. Re-use of sanitary porcelain was found to be sustainable from all aspects, but was not as eco-efficient as brick-cleansing.

The study showed that it was possible to use the triple-bottom-line framework model suggested to make a sustainability assessment of waste management methods. It also showed that the data collection required to perform such an assessment is time-consuming and therefore costly. A conclusion from this was that it would be desirable to reduce the number of indicators studied, if this could be done without impairing the quality of the analysis.

Another important result in Paper I is that it demonstrated that an assessment based only on eco-efficiency indicators would reach different conclusions on which activity is the most sustainable, than one that also includes social aspects. The conclusion that can be drawn from this is that indicators of social aspects must be included in decision support tools for sustainable waste management.

In addition to Paper I included in this thesis, the findings from the first case-study are also described in Klang *et al.*(2001b); Klang *et al.*, (2001a) and Klang (2000).

3.3 Sustainable management of combustible household wastes

The second case-study was a further development of the evaluation and assessment model described in the first case-study. It focused on a project on the management of solid household waste in a sparsely populated municipality in northern Sweden. The main focus was on biological and other combustible household waste, since these are currently given special attention in municipal waste management plans, due to new and forthcoming legislation making it illegal to landfill these wastes.

The aim of the study was to use and further develop the evaluation model to assess the sustainability of different waste treatment methods for these fractions. A further objective was to analyse whether or not national goals and legislation on waste management seem to be relevant to the special conditions in sparsely populated areas. Four treatment methods were chosen as scenarios for the comparison:

- Incineration, with subsequent landfilling of ashes
- Digestion of biological waste, in combination with incineration of the remaining combustible waste.
- Composting of biological waste, in combination with the incineration of the remaining combustible waste.
- Landfilling of all waste, requiring an exception from forthcoming legislation.

As a result of the previous case-study, the number of indicators studied was reduced. From the environmental perspective only emissions contributing to the greenhouse gas effect, acidification and eutrophication were studied. Cost studies were limited to transport and treatment costs, and social aspects to the physical working environment, the level of service for households and the jobs generated within waste management. A functional unit for the study was chosen, and complementary systems that would fulfil the complete functional unit regardless of the treatment method chosen were defined.

In comparison to other studies of waste systems, it was shown that the transportation of the waste and of the replaced fuels would have a more significant impact on the result than in more densely populated areas (Björklund *et al.*, 2000). Thus landfilling appeared to be more beneficial to the environment from an acidification and eutrophication perspective. From a greenhouse gas perspective, however, landfilling was by far the worst scenario. There is, however, a relatively high content of plastics, produced from fossil oil, in household waste. This means that in a distant future, assuming that fossil fuels are phased out of our energy production systems, landfilling would be a better option from a greenhouse gas perspective. Because then waste would replace biofuel when combusted, while the landfill on the other hand could function as a carbon sink. Had an infinite time horizon been used in the study, however, it is probable that most of the fossil carbon would be released from the landfill and the positive effect of landfilling plastics would be reduced accordingly.

Landfilling also proved to be the most expensive treatment method, due to the landfill tax that was introduced in Sweden a few years ago. Without this tax, the landfill option would have been competitive with other methods, from an economic perspective. Other studies have suggested that composting can only be competitive if landfill costs are very high due to the lack of suitable locations for establishing new landfills, or very high land prices (Renkow *et al.*, 1998). This case study has shown that a sufficiently high landfill tax will have the same effect.

From a methodological point of view the most important finding was that the definition of the complementary systems has a large impact on the results. The assessments are complicated by the fact that it is not evident what assumptions to make regarding which fuels would be replaced when waste was incinerated, and what would happen to these fuels. The results generated were presented in a matrix giving a broad overview of the findings, but without providing any basis for comparing the different aspects with one another. It was suggested that such a basis could be developed from a normalisation procedure, as is often done in life-cycle assessment.

3.4 Sustainability of wastewater treatment

This case-study was carried out in co-operation with another project, in which the possibilities of improving wastewater treatment in cold climates by using micro-algae were investigated. In the case-study, a comparison was made between a theoretical waste water treatment plant using microalgae and two other facilities. The ability of the different plants to establish and maintain a path of sustainable development was analysed by two different methods of assessment: emergy assessment and socio-ecological principle assessment. Azar *et al.* (1996) suggested indicators for assessing to what extent a certain activity violates any of the four system conditions for sustainability described by Holmberg *et al.*(1996), and also presented by Robèrt *et al.* (2002) (se chapter 1.2.1).

As it turned out, both the emergy analysis and the sustainability assessment based on the four system conditions came to the same conclusion. This was that the proposed wastewater treatment plant using micro-algae was in a better position than the other two types, to establish and maintain a path of sustainable development.

An important conclusion that can be drawn from the study is that the holistic sustainability assessment methods based on highly aggregated indexes provide results with a lower validity for municipal decision making than the model used in Papers I and II. This is particularly true if the waste management decision makers are committed to considering the economic, environmental and social aspects of waste management.

The results from this case study and the methodologies used have also been presented and discussed in further detail by Grönlund *et al.* (2004b).

3.5 Systems analysis for decision support in waste management

The fourth case study was dedicated to analysing how waste systems analysis was perceived by municipal officers and local politicians, and how valuable it is to the decision-making process. The eight municipalities of Jämtland, Sweden commissioned the systems analysis in question. The main reason for the municipalities having this analysis made, was the forthcoming ban on landfilling of bio-degradable waste, which will come into effect on January 1st, 2005. The firm Carl Bro Intelligent Solutions AB was commissioned to perform the analysis. The analysis was to predict the economic and environmental consequences of four different scenarios, all of which were technically feasible and in accordance with the forthcoming regulations and legislation.

The case-study was carried out through questionnaires that were sent to local politicians and municipal officers. The questionnaire contained questions about the respondents, and in what way they had been involved in the project, which scenarios they preferred and which aspects of the analysis were of greatest importance for them when evaluating the scenarios. The respondents were also asked to indicate to what extent they agreed or disagreed with a number of statements. The solution that both the politicians and the municipal officers preferred was a scenario that suggested that biodegradable household waste would be treated in dispersed composting facilities, while other combustible household waste would be transported to incineration plants outside the county.

To the question about which aspects were of most importance for how the scenarios were valued, the respondents emphasised the possibility for co-operation between the local authorities to minimise costs and negative environmental impacts, sound working conditions for refuse disposal personnel and low emissions of greenhouse gases. Aspects considered of relatively little importance were the number of jobs generated locally and minimising the workload for households. Some differences were noted in how men and women, politicians and municipal officers, respectively, had valued different aspects, but all respondent groups agreed that the possibility of co-operation was one of the most important aspects.

An analysis of to what degree the respondents concurred with different statements showed that, on average, the respondents in this case-study seemed to be content with the systems analysis. Of the six statements that the respondents on average agreed most strongly with, five expressed positive views about the systems analysis. The sixth statement on the ‘top-list’ was the statement that the respondents agreed with most wholeheartedly: ‘One conclusion that can be drawn from the analysis is that national objectives and legislation on waste management are poorly adapted to the special conditions in sparsely populated regions’.

Several respondents expressed a need for further guidance as to how the systems analysis should be interpreted. In this case, the systems analysis was presented as a full extensive report (Leander *et al.*, 2003), which less than half of the respondents had read, but also in a short summary of this report (Bengard, 2003) which a majority of the respondents had read. Several oral presentations had also been given to the municipal officers and local politicians. The responses give no clear indication of what more could have been done to facilitate interpretation, but indicate an interesting and important field for further studies.

An important conclusion from this study is that the decision-makers were interested in decision support regarding all aspects of sustainability in waste management, and the respondents prioritised economic, environmental and social aspects, to be among the most important ones.

4. Analysis and conclusions

4.1 A framework for assessment of sustainable waste management

After the findings from the empirical work described in chapter 3 it is possible to present the basic structures of a framework to analyse different waste management options' potential to contribute to a sustainable development, and how this framework can be used to facilitate such a development. The structure is also presented in Figure 12.

4.1.1 Stakeholder demands

Waste management concerns many stakeholders. Regardless of how well a waste management plan functions in theory, it must gain public acceptance to perform in practise. Waste management planning must therefore always start in dialogue with general public, industry and concerned public authorities to establish a good understanding of stakeholder demands and expectations. After the dialogue, an analysis of the expressed demands is made, and 'critical demands', that must be fulfilled to gain public acceptance, are identified. Local policy and decisions in existing plans already passed by the municipality will be of special importance. Before the next step of the process is initiated, it is also necessary to decide upon a relevant time period. That is, for which period of time should the analysis be valid. One way of determining this is to use the depreciation time for the largest investment in the study.

4.1.2 Choosing scenarios and scenario development

The stakeholder demands form the basis for the scenario development. As a rule, all scenarios should at least meet the basic criteria set by legislative demands and regulations from waste management authorities. If special conditions make it interesting to investigate an option that doesn't fulfil such criteria, it can be done, but it must be made explicit to decision-makers if such a scenario is included. As a reference scenario to which the others can be compared, a null-scenario corresponding to the present treatment system (possible adjusted to fulfil legislative demands) should be included among the scenarios. By determining the comprised 'functional unit' that the system is supposed to deliver (as was done in Paper II), it becomes obvious that each scenario also must contain complementary systems, to enable a just comparison between them.

4.1.3 Determining indicators and collecting data

Indicators covering all aspects of sustainability should be used. Specifically, indicators related to the identified 'critical' stakeholder demands must be chosen, to ensure validity and stakeholder relevance. Examples of possible indicators are shown in table 1. It is then time for data collection, and indicator calculation. This is the most time consuming and work intensive part of the process, but it can be simplified through use of basic worksheet models (such a model was developed for the case study in Paper II).

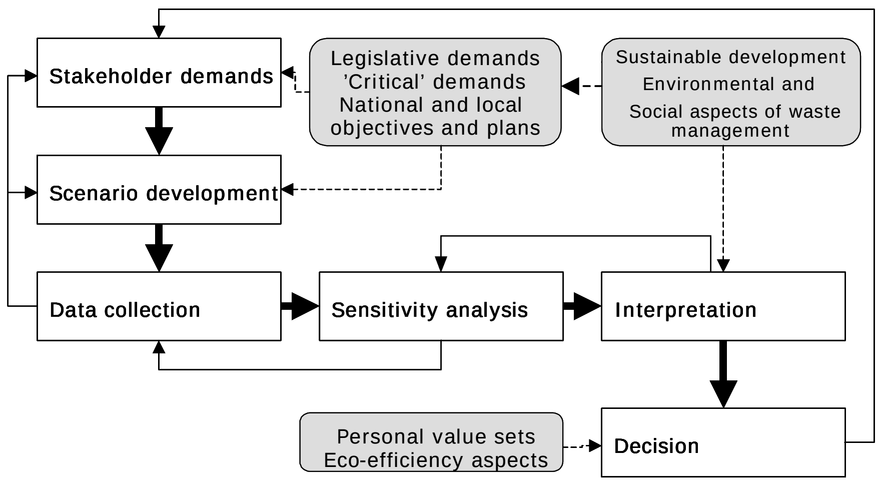


Figure 12. Framework outline. The suggested steps in the iterative chain for sustainable development of waste management systems. Grey boxes indicate important influences on the process in certain steps.

4.1.4 Sensitivity analysis

Once selected indicators describe all scenarios, it is necessary to perform sensitivity analysis, to indicate in which parameter intervals the indicator values are valid. Different data sources will be subject to different uncertainties, and in some cases data with unknown margins-of-error might be used. By varying critical input values within plausible intervals, the reliability of the calculated indicator values can be assessed. A sensitivity analysis that varies the performance, or choice of complementary systems might also be appropriate, either if it is uncertain what the appropriate complementary system should be, or if it is likely that the performance of the complementary system will change over the time period studied.

4.1.5 Interpretation and informed decision-making

Once the intervals of validity are assessed, it is time to interpret the indicator results. To initiate a sustainable development of the waste management system, the following steps can be used in the interpretation:

1. Use ratio-indicators to get a preliminary overview of each scenario, and their relative ‘effectiveness’ in terms of eco-efficiency and socio-economic aspects.
2. Analyse indicators of environmental and social aspects. Only scenarios that have a better performance than the null scenario on these aspects should be considered for further analysis. If the results are ambiguous, normalisation, where the waste management contribution to a certain aspect is compared to the total contributions to that aspect from the municipality (see Paper II), can be applied to separate large impacts from smaller ones. For instance, in a municipality where unemployment is low, the importance of locally generated job opportunities will be reduced.
3. Determine which aspects have the highest priority (if uncertain, normalisation can again be of help), and base the decision on the indicators corresponding to these aspects. Make alternative priorities among the aspects and analyse if that would alter which scenario that appears to be most favourable. If a scenario turns out as the preferable one regardless of which aspects are prioritised, the choice is easy. If not, an informed decision maker must be able to determine the priorities. National environmental objectives and assessments of how far away from reaching these objectives the society is can serve as guide-lines for the prioritisation.
4. Ensure that economical resources are used efficiently. If the costs for avoided emissions are unreasonably high, compared to if similar reductions were obtained through other means, or in other sectors of the municipality, this should be taken into consideration.

By following these steps the framework can promote decision-making towards a sustainable development of a municipality waste management system.

4.2 Validity of the results

It is of course important to analyse the validity of the results presented in the papers. Will the suggested indicator framework actually analyse what we understand sustainable development to be? The key question here is how sustainable development is defined.

The most well known and often cited definition of sustainable development was introduced by the Brundtland Commission (WCED, 1987) and is: “development that meets the needs of the present without compromising the ability of future generations to meet their own needs”. As mentioned earlier, this definition has been subject to criticism, regarding its anthropocentric perspective. In chapter 1.2.2 it is described how later international endeavours have sought to expand the concept and make it more comprehensive.

Another approach could be to simply conclude that sustainable development is development that leads towards a sustainable society. The four system conditions (Holmberg *et al.*, 1996; Robèrt, 2000) described in chapter 1.2.1 is one example of how to define a sustainable society. Figure 4 in chapter 1.2.4 also serves as a good illustration of what constitutes an ecologically sustainable society.

One could say that the four system conditions represent something we could call absolute sustainability. The theory behind the empirical work in this thesis has been that if two systems that fulfil the same function are compared, choosing the system that violates absolute sustainability the least is most likely to lead to sustainable development. This could be referred to as “relative” sustainability. However, it is dangerous to assume that choosing the “less bad” option will eventually lead to a sustainable state. The system in question could very well turn out to be a dead-end (Robèrt *et al.*, 1997), as discussed in chapter 1.3.3.

Nevertheless, there is something very close to a scientific and political consensus that sustainability, and thereby also sustainable development, includes environmental, economic and social parameters (Robèrt *et al.*, 2002; Rogers *et al.*, 2001; Veleva *et al.*, 2001; Elkington, 1997). This is strong indication that the suggested model tested in case studies 1 (Paper I) and 2 (Paper II) as well as the method based on the four system conditions used in case study 3 (Paper III), represent a valid approach to sustainable waste management. This approach may provide different answers (Klang *et al.*, 2001b) than assessment methods that only consider environmental and economic aspects, such as eco-efficiency (OECD, 1998).

The conclusion is that the suggested model does provide a framework for a valid sustainability assessment, *as we understand the concept of sustainable development today*.

4.3 Reliability of the results

An equally important question is whether the method produces reliable results. Can we be sure that the figures obtained are correct and not systematically flawed or erroneous?

A problem with sustainability assessments such as these are that they rely on a large and disparate set of data with varying, and sometimes even unknown, reliability or confidence-intervals. The scientific approach used to deal with this fact is to identify key elements and important uncertainties, and vary them in a sensitivity analysis, and evaluate how this affects the outcome of the analysis.

The sensitivity analysis carried out in case study 2 showed that the results obtained were confident and robust, within the assumed range of variations of the input parameters. Another approach could be to analyse the data set using another model, and see if the results and conclusions differ significantly. This approach has not been used in this PhD-project.

4.4 Relevance from end-user perspective

Just as important as reliability and validity from a scientific perspective, is to what extent practitioners will perceive the suggested methodology as relevant, meaningful and appropriate, both municipal officers and elected local political representatives.

One of the major reasons to carry out the case study presented in Paper IV was to answer the question if a systems analysis is perceived as a valuable decision support tool or not. Unfortunately, no large systems analysis covering more than one municipality, based on a sustainability assessment model, has been made. The case study was therefore carried out among a group of respondents that had been involved in another type of life-cycle based systems analysis (Leander *et al.*, 2003). Even so the case study does provide some helpful indications on the matter.

Respondents expressed very positive views on the usefulness of systems analysis for decision support, and they also had fairly common views about which aspects are of greatest importance when evaluating alternative methods of waste management (Paper IV). Among the most important aspects were environmental, economic and social considerations including:

- Possibilities for municipal co-operation to minimise costs and negative environmental impacts
- Sound working conditions for refuse disposal personnel
- Keeping emissions of greenhouse-gases such as carbon dioxide and methane low
- The economy of the households.

By ensuring that the model includes all these aspects, relevance for the end-user can be secured. Each particular case will require adjustments of the indicator set to provide end-user relevance and manageability, but suggested key indicators, applicable in waste management sustainability assessments, are listed in Table 1. In each specific case indicators may be added or removed from the list, depending on the demands expressed by the stakeholders, as discussed in chapter 2.5

4.5 Principal conclusions from the empirical work

Sustainability is a complex concept, and developing systems analysis support tools for sustainability assessment is therefore also a complex task. Many aspects measured on completely different scales must be considered for an assessment to really fulfil the criteria for sustainability assessment, and the resulting outcome might be confusing and difficult for the decision makers that would commission such assessments to fully understand. The work carried out in this project has shed some light on these difficulties, and pointed to possible solutions to some of them. It has been shown that it is possible to present comprehensive results in a form that facilitates interpretation, without providing a yardstick by which all aspects should be measured.

From the papers included in this thesis, some very important conclusions can be drawn:

1. Municipal decision-makers prioritise both environmental and economic as well as social aspects when they evaluate waste management options (Paper IV).
2. Municipal decision-makers and municipal officers consider a holistic systems approach preferable to one that is based only on costs or only on the environmental effects (Paper IV).
3. A method considering social aspects together with economic and environmental ones may lead to different conclusions than one only considering eco-efficiency (Paper I).
4. A systems analysis is very dependent on the definitions of complementary systems, and a thorough investigation of possible technological developments is crucial for a reliable analysis (Paper II).
5. Methods generating results on a very aggregated level, without ensuring a full life-cycle perspective, or omitting one or more aspects of sustainability are not valid for waste management decision making (Papers I, III and IV).
6. A wide-spread opinion among local decision makers and municipal officers, that national directives and objectives for waste management are poorly adapted to the conditions of sparsely populated regions (Paper IV), does not seem to find support from systems analysis using sustainable development indicators (Papers I and II).

One way to help decision making based on systems analysis, which is often used in life cycle analysis studies, is to assign weight to all the aspects included, and thereby derive a final variable that summarises all the effects. There are many different methodologies suggested for doing this, perhaps among the most notable Eco-scarcity, EPS and Ecological Footprints, all developed to measure different environmental effects on a common scale. There are also methods calculating all the effects in monetary terms, again making it possible to measure everything in one variable. All of these methods, however, have in common that they are based on underlying, and not always evident, assumptions and value-sets. The question is how far an assessment model should go, and at what point it must be up to decision-makers to take responsibility for the decision?

The conclusion from the empirical work that this thesis is based on is that decision makers as well as municipal officers have a clear view of what aspects of waste management they regard as most important. What they therefore need is not models that provide a finite answer, but assessments that describe the consequences of all aspects as transparently as possible, which allows individuals with different value-sets to draw their own conclusions. What the sustainability assessment should provide, however, is guidance in interpreting the results, depending on which aspects are given the highest priority.

4.6 Need for further research

The presented model is far from being a completed tool that could be set in the hands of practitioners to work with immediately. Guidelines need to be developed to instruct waste management planners on how to decide which aspects and associated indicators to utilise in each assessment.

Data sets with up to date life cycle based environmental data must be made available, and systems to keep these data sets current must be in place. Technologies will improve over time, and new treatment methods will evolve. The spreadsheet-based computer model developed for Paper II serves as a good starting point for this purpose, but needs to be developed further, using other case studies and examples, before it can be re-used for generic, rather than case-specific, waste management modelling.

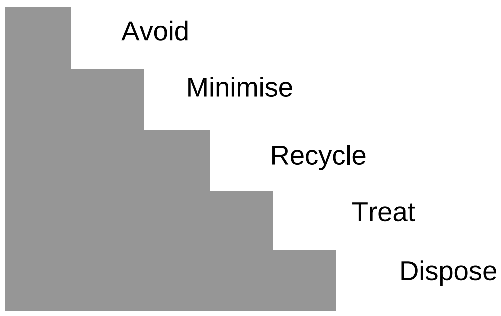


Figure 13. The European Waste Management Hierarchy, first introduced in the Second Environmental Action Programme (1977-1981)

As our understanding of what a sustainable society really means evolves over time, so must our methods to assess sustainability and sustainable development evolve as well, and different indicators than the ones suggested might come more in focus. But more important than evolving technologies and shifting focus in the sustainability discussion, is the fact that waste management so far has been unsuccessful in dealing with the first steps in the so called waste-hierarchy (Wilson, 1996), namely source reduction to avoid or minimise the production of waste to begin with (Figure 13).

Initially, some success was noted in Sweden after the introduction of EPR on packaging with lighter and smarter designs (Tiliander, 1998). But the role of the packaging is only partially to protect the product. Equally important is to attract the customer and convince them to choose the product contained, and sticking out with an elaborate and colourful packaging is certainly one way to achieve this.

To really reach success in avoiding and minimising waste generation, the consumers must be more involved and informed in waste management issues. Future research in waste management must address how this can be achieved, and how transparent and systematic waste management assessment models may be used as a platform for communication to the public..

4.7 Future waste management in sparsely populated regions

At an international workshop held outside Stockholm in April 2001 (Sundqvist *et al.*, 2002), it was concluded that the prioritisation of the waste-hierarchy is valid as a rule of thumb when considering environmental effects. The thoughts behind current national waste strategies in Sweden are influenced by the waste-hierarchy. The primary focus is on redirecting wastes that are landfilled (enforced and upcoming bans), and to encourage recycling through EPR.

The research carried out in this project has given a preliminary indication that there is a well spread opinion among local decision makers and municipal officers, that Swedish national directives and objectives for waste management are poorly adapted to the special conditions of sparsely populated areas. Regardless if this opinion is based on facts supported by scientific findings, or merely on a general conviction based on a gut feeling that the increased transports will outweigh environmental benefits from source separation and decreased landfilling, something will have to be done to address the issue. Three main strategies can be identified;

1. Carry out further studies to determine if the general rules of the waste hierarchy are valid in sparsely populated areas.
2. Continue to develop available technologies that are potentially competitive regardless of treatment volumes, such as composting, to reduce the need for transportation.

3. Develop new technologies suitable for treatment of smaller volumes of waste (e g small and medium scale incineration plants), and benefits from the positive factors in rural and sparsely populated areas (low land costs, large and sparsely populated land areas).

Another possible measure that could mitigate negative environmental influence from waste management in sparsely populated areas, could be to co-ordinate waste collection or collection of source separated materials with other transports to remote areas.

4.8 Concluding remarks

Waste management is indeed important for our ability to reach a sustainable society. We need improved practices in order to close necessary material loops, and we need to ensure that toxic substances are not released to the environment. But far more important for sustainability, are the unsustainable consumption patterns that we have today. These patterns are responsible for the world wide increase in waste generation, and improving waste management will not rectify the problem, only mitigate it's consequences.

Figure 11 indicated that we can position our present condition as the origin of a system of co-ordinates, and that sustainability is found in the far quadrant in the positive side of this system. However, if we consider the 'Economic aspects' axis to represent wealth and assets, in is clear that on a global level, we have enough wealth and assets. The problem is their uneven distribution. In many countries today, the richest 10% of the population have a percentage share of income or consumption close to 50%, while the poorest 10% sometimes are responsible for less than 1% of the consumption (World Bank, 2002). Differences between rich and poor countries are in some cases even larger. Moving development further along the economy axis, without consideration for environmental or social aspects will therefore in reality not move us any closer to a sustainable society.

Rural and sparsely populated areas may appear to have certain difficulties to transform their waste management systems in a more sustainable direction. Small waste volumes, long collection routes and distant treatment facilities are some factors that have been mentioned in this thesis. But simultaneously, there are favourable factors as well.

In rural areas farmland is always close at hand. Returning nutrients to productive soil will be one of the most important tasks for future waste management. Rural areas can of course play an important role in the front of this development.

Large areas, and sparse population are also advantages when looking for suitable locations for treatment facilities. Low land cost prices can make it economically advantageous to apply low intensity solutions (Paper III), using ecological engineering techniques to treat waste water and solid organic waste close to the source, with minimal inputs of fossil fuels or chemicals. Again, sparsely populated areas have a better position to apply such solutions.

On the whole, one has to acknowledge that sustainable waste management in sparsely populated areas will not, most likely, be organised in the same manner as in urban environments. But the possibilities to achieve a sustainable waste management are by no means smaller in such areas.

Table 1. Examples of generally applicable indicators for use in sustainability assessment of waste management methods.

Indicator type	*Topic of interest*	*Examples of indicators describing the topic*
Environmental [a]	Climate change	Emissions of CO_2-equivalents
	Ozone depletion	Emissions of CFC11- equivalents
	Acidifying potential	Maximum theoretical H^+ formation
	Photochemical Ozone Creation Potentials (POCP)	Emissions of ethene-equivalents
	Eutrophication potential	Maximum theoretical O_2 demand
	Land use	Total area of treatment facilities
	Energy use [b]	Total energy use Use of fossil fuels Use of electricity
	Resource use	Throughput of renewable resources Throughput of stock resources
Economic	Operational costs (OC)	Labour costs (including costs for work related sick leave) Collection costs Treatment costs
	Investment costs (IC)	Yearly depreciation costs
	Revenues (R)	Revenues from selling reused goods or materials to recycling
Social	Physical work load per employee	Assessment of physical working conditions
	Health and security issues of physical work environment	Air-quality assessment regarding microbiological contaminants and gaseous compounds.
	Psycho-social work environment	Assessment of psycho-social working conditions
	Level of service	Number of fractions in curb side collection.
		Average distance for households to collection point for non curb side fractions.
Eco-efficiency	Environmental influence per total cost	Each of the categories under environmental indicators divided by the total cost.
	Environmental benefits/hour of labour	Avoided emissions and energy consumption divided by hours of labour within the waste management system.
Socio-economic	Total employment	Yearly number of labour hours.
	Health-care and sick leave costs	Estimated societal cost for work related disorders and diseases that arises within the system.
Environmental awareness	Public participation	Percentage of correctly sorted materials in different fractions.
		Customer satisfaction with level of service (pleased to very pleased) as reported in questionnaires.
	Willingness to pay	Self-reported willingness to pay for environmental improvements of waste treatment.

[a] All suggested indicators include emissions, resource or energy consumption in new production to substitute materials and goods not recycled or reused within the system

[b] Emissions from energy use are included in the indicators referred to above.

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Sustainable management of demolition waste—an integrated model for the evaluation of environmental, economic and social aspects

Anders Klang [a,*], Per-Åke Vikman [a], Helge Brattebø [b]

[a] *Department of Natural and Environmental Sciences, Division of Ecotechnics, Mid-Sweden University, SE-831 25 Ostersund, Sweden*

[b] *Programme of Industrial Ecology, The Norwegian University of Science and Technology, Gløshaugen. 5, N-7491 Trondheim, Norway*

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Abstract

A model is presented for evaluating waste management systems for their contribution to a sustainable development, including environmental, economic and social aspects. The model was tested in a case-study, where groups of long-term unemployed people were offered both education on environmental issues and practical work with the recovery and recycling of building and demolition waste as a form of vocational development. Application of the suggested model revealed the overall effects on sustainability of different methods of waste management. In addition, negative aspects of the systems analysed were identified, which led to discussions about possible improved practices within the waste management systems. Two of the waste management systems investigated (the recycling of steel and re-use of sanitary porcelain) showed a potential contribution to sustainable development in all of the aspects studied. Preparing bricks for re-use showed the largest potential for eco-efficiency, but had negative effects on sustainability from the social perspective of health and the working environment. The possibility of further use of the model and the remaining obstacles to such analyses are discussed. One observation is that the data collection needed to perform this kind

* Corresponding author. Tel.: +46-63-165760; fax: +46-63-165-450.
E-mail address: anders.klang@mh.se (A. Klang).

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of sustainability analysis is resource-demanding, and that it would therefore be better to identify a smaller number of key indicators.

Keywords: Construction and demolition waste; Life-cycle analysis; Triple bottom line; Sustainable jobs; Unemployment

1. Introduction

1.1. The concept of sustainable development

Internationally, sustainable development has become a well-known concept through the work of the World Commission on Environment and Development and their report *Our Common Future* (WCED, 1987). Their often cited definition of sustainable development focuses on our obligation to ensure that future generations abilities can meet their needs, but also to work towards a more equal distribution of wealth at present. These, and other aspects of sustainability, were also included in the declarations from the UN conference in Rio de Janeiro in 1992 (United Nations Conference on Environment and Development, 1992). The *Agenda 21* endorsed in Rio suggested that methods for monitoring trends of sustainable development needed to be developed, and particularly emphasised the need to integrate environmental accounting with traditional macro-economic calculation methods.

This may be one reason why the main focus of the scientific debate on sustainability in the last decade has been at the macro-level. The objective has been to describe how entire communities, or even nations, are developing towards or away from sustainability (Rees and Wackernagel, 1994; Anonymous, 1999). This is important, but there is also a need to develop methods to evaluate sustainability and sustainable development on a smaller scale, such as in businesses or projects (Read, 1999). One reason for this is that it is not always obvious how individuals in their respective countries can effect progress towards sustainability if it is only approached at the macro-level. Setting up targets in a limited system can be a more efficient way to influence behaviour, and thereby also contribute to fulfilling sustainability goals at the macro-level (Dwyer and Leeming, 1993). In conclusion, to claim that a local activity is sustainable, or leads towards sustainability, environmental, economic and social issues must be taken into consideration.

1.2. Methods of measuring sustainability on a smaller scale

Over the years, several efforts have been made to find means for companies to integrate other aspects than strictly economic ones into their accounting systems. To do so, it is necessary to identify other, non-traditional values, and other resource-bases that are essential to the operations. One example of such a method, is Sustainable Development Records (Nilsson and Bergström, 1995; Nilsson, 1997). The main objective of the organisation World Business Council for Sustainable

Development (WBCSD) is to achieve wider corporate responsibility (without a reduction in profits). Their work has focused largely on the concepts of cleaner production (WBCSD and United Nations Environment Programme, 1998) and eco-efficiency (WBCSD, 1999). Several methods for the development of indicators of sustainable development that can be used in projects or geographical areas at different levels have been described (Mitchell, et al., 1995; Pinter et al., 1995; Kuik and Verbruggen, 1991). Other initiatives have had similar aims, such as the concept of 'triple bottom line'-accounting (Elkington, 1997), which is also designed for making small scale analyses, emphasising the need for businesses to include social and environmental dimensions in their performance reporting and in implementation of corporate business strategies (Movat, 2002). The main reason for this is that today, wider groups of stakeholders (including employees, NGOs, local communities and perhaps most importantly customers) demand that companies disclose information about their over all impact (Hedstrom and Isenberg, 2002), whereas company performance reporting in the past was mainly of interest to stockholders.

Veleva and Ellenbecker (2001) have formulated a set of principles to describe what they regard as Sustainable Production (Veleva et al., 2001b). These principles include products and services, processes, working conditions, relations to communities and economic viability. Indicators to monitor a company's compliance to the principles are under development, and core indicators, applicable to all types of production, have been suggested (Veleva and Ellenbecker, 2001) and tested (Veleva et al., 2001a).

1.3. Measuring sustainable development at intermediate levels

As shown above, there are a number of tools available for assessing sustainable development, both at the national level and at a company level. There is, however, a need to develop better tools to measure sustainable development at an intermediate level. Such tools could, for instance, be used by municipalities, enabling them to link results on corporate and project levels to sustainability targets for a community or the entire nation. Efforts to develop tools to promote sustainable cities have been made (Priemus, 1999; The Swedish Research Council, 1995), but the model suggested in this paper focuses on the sustainable management of waste.

1.4. Unemployment, ergonomics, recycling and sustainability from a social equality perspective

Unlike many other European countries, Sweden maintained very low unemployment figures throughout the 1980s. In the beginning of the nineties this changed dramatically, when a slowing economy caused many businesses to collapse or at least reduce their work forces. Construction almost came to a complete stop, and unemployment rose to unprecedented numbers in only a few years (Swedish National Labour Market Board, 1999). This made it increasingly difficult for young people to access the labour market. Studies have indicated that there is a correlation between unemployment and nervous and depressive symptoms among youths

(Hammer, 1993; Hammarström and Janlert, 1997). However, studies have shown that not all the jobs created in the waste collection and recycling business are sustainable from a health/working environment perspective. Employees working with refuse collection and disposal in Sweden are subject to risks of work injuries that are about 3 times that of working life in general (Swedish National Board of Occupational Safety and Health, 1998b; Nordin and Bengtsson, 2001). Neither unemployment nor unacceptable working conditions are socially sustainable, which makes it important to address both these issues when working with sustainable development within waste management.

1.5. Waste management and sustainability from an environmental perspective

In many respects, achieving ecological or environmental sustainability is closely linked to the manner in which we deal with the waste products of society. Visions of what constitutes an ecologically sustainable system for waste treatment have been suggested (Tiberg, 1993). It is also apparent that the mass of waste products released to the atmosphere as 'molecular-waste' in the industrialised countries, greatly exceeds the amount of solid waste generated per capita. A study, made by the World Resource Institute of material flows in a number of industrialised countries, showed that one half to three quarters of the annual material input to these societies was returned to the environment as waste within a year (Hutter, 2000). Even so, closing material cycles by re-use and recycling does not necessarily result in ecological sustainability. Sometimes the effort to recycle may in itself cause a severe environmental impact. For this reason it is important to maintain a life-cycle-perspective when evaluating waste management measures. Life-cycle-assessment is a tool that has developed rapidly over the last years and international standards are developed (International Organization for Standardization, 1998). The life-cycle-perspective is also stressed in the draft standard (technical report) for Environmental Product Declarations, ISO TR 14025, and in national versions of this technical report (Swedish Environmental Management Council, 2000). As the number of goods with standardised/certified environmental product declarations increases, it will become possible to compare the costs and benefits of recycling and re-use with the potential effects of the production of corresponding goods from virgin raw materials.

1.6. Sustainability and economic aspects

The construction and demolition sector deals with a number of hazardous materials, for instance asbestos mats, PCB-contaminated joint compositions, CFC-gases in cooling installations and many others. Such materials and substances have to be taken care of properly, regardless of costs. To ensure this, it is necessary that adequate legislation is passed. However, companies will always seek to maximise profits, and in order to promote re-use and recycling beyond the legislative demands, it is often necessary to point out the economic benefits of such operations. Alternatively, instruments of control can be enforced that make environmentally optimal alternatives also economically optimal from a business perspective. Landfill taxes are an example of such a measure (Morris et al., 1998). A company can of course find that an activity generates other values than strictly economic ones, or that it generates income indirectly, and therefore choose to perform it anyway. In this study, however, we have chosen to regard the situation as economically sustainable when an activity generates an income that is equal to, or preferably larger than, its costs.

1.7. The case at hand

For 2 years the municipalities of Steinkjer and Trondheim in Norway, and Östersund in Sweden have been co-operating in a project that aims to re-introduce long-term unemployed people to the labour market. The aim of the project was threefold: (i) to describe how the environmental impact of the building sector would be influenced by increased recycling and re-use of demolition material; (ii) to contribute to the social competence of the participants and increase their chances of finding work in the future; and (iii) to discover if it is possible to identify certain activities as long-term 'sustainable green jobs', even from an economic perspective. A common interest was identified: to work with issues related to social competence and environmental knowledge as tools to achieve vocational development. Another mutual interest was to develop a tool to identify 'green sustainable jobs', i.e. jobs that have a beneficial impact on the environment, provide a physically and psycho-socially sound working environment and that generate a large enough revenue to cover salaries and social costs. The building and demolition sector could provide suitable objects to work with in all three municipalities.

The building and demolition sector is a major source of solid waste generation in both Sweden and Norway, as well as in many other European countries (Symonds Group Ltd, 1999). In the mid-nineties Swedish contractors and building material producers therefore agreed on a voluntary extended producer responsibility. One of the objects was to decrease the amount of waste brought to landfills, from construction and demolition sites. Measures and a time plan to achieve this and other goals were specified in an action plan (The Ecocycle Council for the Building Sector, 2000). An important part of the action plan deals with the subject of education on environmental issues and selective demolition. The Swedish participants in the project therefore initiated a partnership with the local representatives of the Eco-cycle Council for the Building Sector regarding specialised education on selective demolition techniques etc. A programme for environmental education was developed in co-operation with Mid-Sweden University. The practical work then took place in periods of 6 months on a major construction site in Östersund. Two groups of four employees each worked within the project. In total, 10 persons were employed, but two left the project soon after starting, without filling in the questionnaires used in the study (see below), as these were only given to the persons replacing them.

1.8. Objectives and limitations of the case-study

The objective of the case-study was to investigate whether or not it is possible to construct an evaluation model to compare multiple aspects of sustainability in different recycling and re-using activities. The study was restricted to comparing the effects that can be measured at the construction or demolition site and during the transport of goods and materials to the final receiver, to the effects reported in life-cycle analyses of construction materials made from virgin raw materials. The studied recycled or re-used products were assumed to meet consumer quality demands regarding performance and life expectancy.

2. Methods

2.1. Model design

A model for data collection was designed, based on three key aspects of the project from a sustainability perspective. The three aspects were environmental aspects, economic aspects and social aspects. A set of indicators was selected, based on discussions with stakeholders in each participating community. From these, a smaller number of generally applicable indicators were singled out, based on their relevance to sustainable development and the possibility of obtaining reliable data. After a final discussion with the stakeholders, these indicators were finally established.

2.2. Environmental data

Environmental data were collected through literature studies regarding the life-cycle of building components and products (Erlandsson, 1994; Tillman, 1996). These data were compared to measurements of energy consumption and calculated emissions in the recycling and re-use activities studied. Emissions and energy consumption from the transporting of recycled goods to a retailer were also included, using emission data from the Swedish National Road Administration (Johansson, 2000).

2.3. Social data

Social data were primarily collected through questionnaires regarding both the physical and psycho-social working environment. These questionnaires were given to all the workers participating in the study. Respondents were asked to give their own subjective assessment of eight different aspects of the psycho-social working conditions on a scale from 1–5 (Fig. 2). All the workers were also asked to assess the physical and ergonomic working conditions in 10 different activities, by rating them on a four-level scale from ‘highly unsatisfactory’ to ‘very satisfactory’. The physical work environment was also assessed on an ergonomic matrix (Swedish

National Board of Occupational Safety and Health, 1998a). This assessment divides different tasks into one of three groups: red, yellow or green. If a task is found to be in the red area, it is unsuitable from a musculo-skeletal disorder perspective. If it is estimated to fall in the green area, it is acceptable. Tasks judged to fall in the yellow area need to be more closely analysed and evaluated (Swedish National Board of Occupational Safety and Health, 1998a).

2.4. Economic data

The economic data collection consisted foremost of time-measurement of different work assignments and a follow-up of the costs of the energy consumption of tools and machines, as well as of documented transportation. Other methods used were market price studies, of both recycled (ByggIgen, 2001) and newly produced construction materials, to obtain the approximate economic value of the goods produced.

2.5. Ratio-indicators

By relating indicators from different aspects to each other, different types of ratio-indicators were obtained, which facilitated a comparative study of different activities within the project. When environmental aspects are linked to economic aspects, indicators of eco-efficiency are obtained (OECD, 1998). For example; the ecological indicator ‘kilograms of CO_2-equivalents avoided per square-meter of brick wall when using old cleansed bricks instead of producing new ones’, is linked to the economic indicator ‘number of labour hours needed to produce bricks that will build 1 m^2 of brick wall’. Thereby a figure on ‘kilograms of CO_2-equivalents avoided per hour of work cleansing old bricks’ can be calculated. This operation allows for a comparison of different activities and thereby shows where to concentrate the work effort in order to get the largest possible positive environmental result. Therefore this type of ratio-indicator is regarded as the most important one for this case-study.

A type of socio-economic ratio-indicator can be obtained by linking indicators of social aspects to economic issues. For example; what is the societal cost of having a person unemployed compared to employing a person to work with an activity such as brick cleansing? This is of course dependent on the economic aspects of brick cleansing, and whether or not the activity will generate an added value which can cover or exceed the labour costs, but also on how sustainable the work in question is from a social perspective, including the health effects of the working environment.

The stakeholders in the case studied were primarily interested in re-introducing people to the labour market. For this to succeed, there has to be a demand for labour with the kind of skills provided through the project, and for the type of products generated by the activities. This was monitored by the third kind of ratio-indicators, which consist of indicators that can evaluate to what extent environmentally oriented skills can increase long-term employment. This data is not yet available from the present case-study.

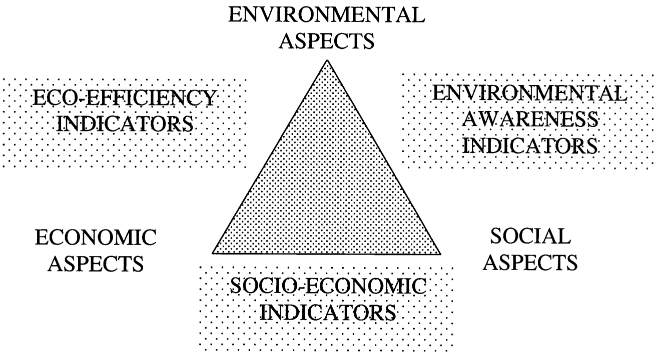


Fig. 1. The categories of indicators in the suggested model.

To fully evaluate the environmental benefits of the activities studied, it is desirable to monitor the reasons for purchase of the customers who buy recycled and re-used products. Only if the re-used products have replaced new products, made from virgin raw materials, can one calculate a genuine environmental benefit. No complete investigation of this type has been performed within this project, but some available data are presented. The categories of all the indicators used in the evaluation model are illustrated in Fig. 1.

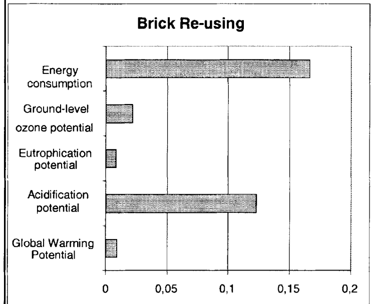
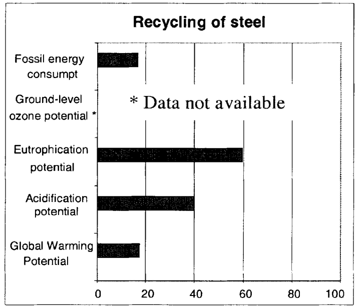
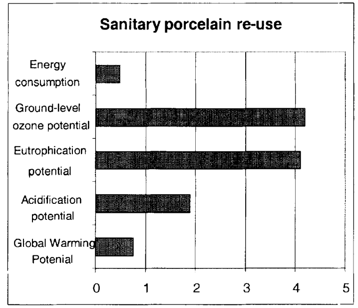
3. Results—the case-study examples

3.1. Environmental aspects

3.1.1. Environmental aspects of re-using bricks

One problem with recycling bricks is that they sometimes do not measure up to modern quality and environmental standards for resistance to crack-formation and thermal conductivity. In this case, however, the bricks were taken from inner-walls, and re-used in other inner-walls in the same buildings, and thereby these problems can be disregarded. To clean the bricks from old mortar, an electrically-powered hydraulic machine with steel edges was used. This machine, manufactured by KomServ in the Municipality of Halmstad in Sweden, was operated by one or two people. Bricks were brought to the machine by a third person and a fourth person stacked the cleansed bricks on pallets. The electricity consumed for cleansing 42 bricks, corresponding to one m^2 of brick wall, with the machine in question amounts to 0.84 MJ. Using LCA-values for Swedish average electricity (Swedish Environmental Research Institute, 2000), the emissions per metre square of brick wall were calculated. These emissions were classified in impact categories and characterised, using internationally accepted weighting factors (Swedish Environmental Management Council, 2000). These values are compared with life-cycle data from primary production of new bricks (Erlandsson, 1994) in the diagrams in Table 1, showing the relative environmental effect from re-using bricks, compared to producing new ones.

Table 1
Overview of different types of indicators in the case-study evaluation. Other indicators were also used

INDICATOR TYPE	*Aspects of the re-use of bricks*	*Aspects of steel recycling or the re-use of steel products*	*Aspects of the re-use of sanitary porcelain (toilets and washbasins)*
Environmental aspects *Environmental load from recycling is expressed as percentage of environmental load from production of corresponding goods from virgin raw material. Please note the different scales.*			
Economic aspects USD 1 = SEK 9:20	Total cost of cleansing bricks covering one m^2 of brick wall **SEK 73:60** Labour costs per brick and current market value excluding VAT of reused bricks, Bygglgen, Sweden. **SEK 1:75 SEK and 8:00** Retail price new bricks, excluding VAT **SEK 6-7:-**	Total costs per unit of steel product **SEK 127** Price on reused goods, at BegMa in Östersund and scrap metal prices paid by Stena Metall **SEK Washbasin 500:- or 3-4:- per kg scrap metal** Retail price new steel washbasin, excluding VAT **SEK 2700:-**	Total costs per toilet or washbasin. **SEK 25:-** Price on reused goods excluding VAT, at BegMa in Östersund (SEK) **Toilet: SEK 520:-, washbasin SEK 200:-** Retail prices new toilets and wash basins, excluding VAT (SEK) **Toilet: SEK 1475:-, washbasin SEK 875:-**
Eco-efficiency ratio-indicators	Energy consumption avoided per hour of labour when cleansing bricks **3300 MJ/h** Kg of carbon dioxide equivalents saved per hour of labour when cleansing bricks **230 kg CO_2-eql/h** Potential acidification (mol H^+) avoided per hour of labour when cleansing bricks **48 mol H^+/h**	Energy consumption avoided per hour of labour when recycling steel products **1600 MJ/h** Kg of carbon dioxide equivalents saved per hour of labour when recycling steel products **86 kg CO_2-eql/h** Potential acidification (mol H^+) avoided per hour of labour when recycling steel products **16 mol H^+/h**	Energy consumption avoided per hour of labour when dismantling toilets. **3900 MJ/h** Kg of carbon dioxide equivalents saved per hour of labour when dismantling sanitary porcelain. **180 kg CO_2-eql/h** Potential acidification (mol H^+) avoided per hour of labour when dismantling toilets. **27 mol H^+/h**
Social aspects	Workers' evaluation of physical working environment during brick cleansing . Average and median response value. **2,6 and 3 (satisfactory)**	Workers' evaluation of physical working environment during dismantling of sanitary ware / equipment of steel and porcelain. 1= very unsatisfactory, 4= highly satisfactory. Average and median response value. **3,1 and 3 (satisfactory)**	
	Workers estimates of socio- psychological work environment during the project (shortly after start-up and close to finishing the project) Eight different aspects measured on a scale from 1-5, where five always is the most positive. **See figure 2.** Percentage of workers that after the project would consider working within the field of selective demolition / reusing of building materials (maybe – would really like to) **100 % (only five respondents)** Degree of employment after finishing project. **30%** Percentage of workers that have continued on to further education after finishing project. **20%**		

As shown, the environmental impact of re-used bricks is only a very small fraction of the potential impact of primary production, and can thereby be said to be more sustainable from an environmental perspective. Since the bricks in this case were re-used in the same building complex as they were taken from, no transport was necessary. However, the total amount of CO_2-equivalents avoided through the re-use of bricks in this project would allow for road transport of 15 000–20 000 km, depending on the vehicle used, before the emissions would exceed those from new production (Johansson, 2000).

3.1.2. Environmental aspects of recycling steel

In this study, stainless steel equipment from showers and toilets was dismantled and prepared for re-use. The potential interest among consumers for buying these products for re-use is uncertain, but if they are not sold as re-use products, they could be sent to recycling instead. A comparison between the environmental effects if these goods were to be recycled, and new production of steel from virgin ore has therefore been made. This can be viewed as a 'worst-case' scenario, since it is very likely that at least some of the stainless steel equipment would be sold as re-use products. The environmental impact of both recycling steel and primary production was derived from Tillman, 1996. The environmental impact of transportation in this case was added using emission data from the Swedish National Road Administration (Johansson, 2000). The second diagram in Table 1 illustrates the relative environmental effect of producing steel from scrap, compared to primary production of steel. As shown, the environmental impact in all the aspects studied is considerably lower for recycling.

3.1.3. Environmental aspects of re-using porcelain sanitary ware

The environmental effects of primary production of sanitary ware was derived from literature and relates to emissions from transportation (Johansson, 2000) of the dismantled goods. The results are illustrated in the third diagram in Table 1, and show that the transportation of the sanitary ware to a retailer of re-used construction goods only represents a few percent of the environmental impact from new production. The real difference is even bigger, since the use of newly produced goods would include a further environmental impact from transportation, which is not considered in the calculations. Re-using sanitary goods can therefore be regarded as environmentally sustainable.

3.2. Economic aspect

3.2.1. Economic aspects of re-using bricks

Two hundred and seventy-five bricks/man-hour could be cleansed by this method, including the time for collecting the bricks from the demolition site and transporting them in containers on wheels to the cleansing machine. That corresponds to approximately 6.5 m^2 of brick wall (again 42 bricks per m^2). In this project, the cost per man-hour, including salaries and social costs, amounted to SEK 120. The value of the re-used bricks can either be obtained from firms specialising in selling re-used building materials, or by comparing with the price of new bricks. In Sweden, this value has been found to vary between SEK 8.30 and 10 per brick (with the higher price for re-used bricks that happen to be in demand for aesthetic reasons). This means that the generated surplus value of the re-used bricks definitely exceeds the labour cost, with a considerable margin. The revenue would be approximately SEK 2000 per man-hour, which would cover the additional costs of storage and machinery and still leave a considerable profit. Brick cleansing under these conditions is thereby shown to be economically sustainable. Even with higher salaries, on a par with those of regular construction workers (Swedish Building Workers Union, 2001), brick cleansing would still be economically sustainable. The labour cost would then be closer to SEK 225 per hour.

3.2.2. Economic aspects of recycling steel

In the present study, time measures regarding the dismantling of multiple tap washbasins, of a length of three meters, were made. The cost of dismantling one washbasin of this type amounted to SEK 120 (equal to one person working for 1 h). The estimated retail price for such a washbasin is SEK 400 excluding VAT, which is well below the cost of a corresponding new product. Even if this low figure is used, the revenue exceeds the dismantling cost with a considerable margin. If the washbasin was to be sent to metal recycling instead, the revenue would only be approximately SEK 115 excluding VAT, which does not cover labour costs. The exact figure is hard to give, since the price of scrap stainless steel varies between SEK 3 and 4 per kg. It is possible that the dismantling time could be reduced if it was clear that the basins were to be recycled and not re-used, so it might be possible to keep the labour cost within this margin, but not if normal construction worker salaries are applied. However, one must also consider the alternative cost, if the material was not re-used or recycled, which in this case would be the landfill fee. In this case, the cost for this would be approximately SEK 25 excluding VAT per steel washbasin and the dismantling time would be more or less the same as for recycling. The level of

Table 2
Economic aspects of the different activities

Activity	Brick cleansing	Steel recycling	Sanitary porcelain reuse
Labour costs[a] (SEK)	73:50	120	20
Energy costs[b] (SEK)	0:15	–	–
Transportation costs[c] (SEK)	–	6:95	4:70
Total sum	73:65	126:95	24:70
Labour costs as percentage of total	> 99%	95%	81%

[a] Per metre squared of brick wall, and unit of steel product (washbasin) porcelain ware (toilet).

[b] Energy costs during dismantling and refurbishing, per the same units as above. Only given for brick cleansing, where an electrically-powered cleansing machine was used.

[c] Transportation costs, including costs for loading, fuel and labour cost for driver. Not given for brick cleansing since they were reused on location.

economic sustainability is completely dependent on the demand for recycled goods of this type.

3.2.3. Economic aspects of re-using porcelain sanitary ware

The dismantling time for porcelain toilets and washbasins was studied and found to be approximately 10 min, costing SEK 20. Both washbasins and toilets are in relatively high demand on the used goods markets, and the sanitary ware from this project was sold at very attractive prices. Toilets were sold for SEK 520 and washbasins for SEK 200, both excluding VAT. There is thereby a considerable margin, making this activity well within the limits of being economically sustainable.

3.2.4. Economic aspects, all activities

In Table 2, an overview of the economic aspects of the activities studied is given. The dominating costs in this case were found to be those related to labour. Costs for energy use and transportation (fuel and labour costs for the truck driver) were found to be less than 1% in the case of brick cleansing, but as much as 19% for the re-use of sanitary porcelain. The figures given are based on the assumption that an optimally loaded light truck, using diesel for fuel, is used. Current local prices for electricity (Jämtkraft AB, 2002) and diesel (Statoil AB, 2002), including taxes, are used. These figures are very dependent on the specific conditions of the case, but it should be noted that bricks can be transported long distances before the environmental benefits in the form of reduced energy consumption would be exceeded (e.g. the CO_2-emissions avoided by laying just 1 m^2 wall of re-used bricks, instead of newly produced ones, equals a fully loaded heavy truck travelling 37 km).

3.3. Social aspects

3.3.1. Social aspects of re-using bricks

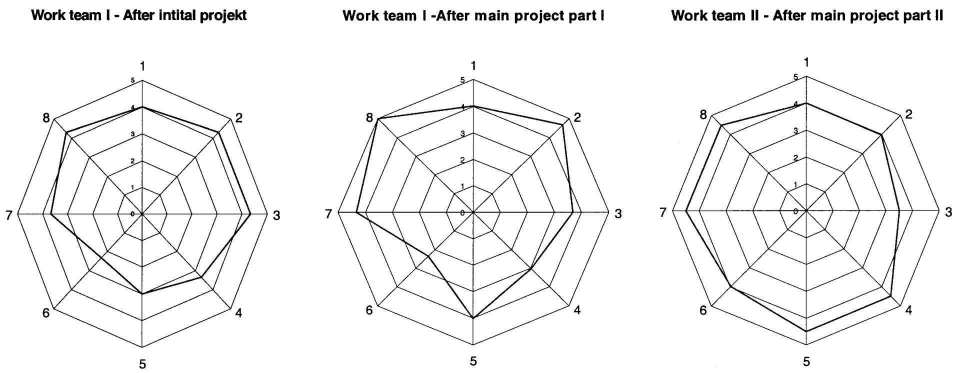
To determine different aspects of social sustainability in different activities, questionnaires were filled out by and interviews made with the participants. A majority of the respondents subjectively rated the physical working environment during one or more phases of the brick cleansing procedure as ‘unsatisfactory’ or ‘highly unsatisfactory’. The main reason was that the individual bricks of this older type weigh too much. According to recommendations given in the regulations (Swedish National Board of Occupational Safety and Health, (1998a)), if the bricks can be handled with only one hand, bricks weighing more than 3 kg are normally considered unsuitable, but these bricks varied in weight between 4 and 5 kg. Thereby, according to the musculo-skeletal matrixes drawn up by the Swedish National Board of Occupational Safety and Health (1998), some phases of the brick cleansing would be classified as being in the ‘yellow area’, calling for a closer evaluation. As a result of this, a number of measures to improve the physical working environment have been suggested by the project management, to ensure that all phases comply with or exceed the demands made in relevant legislation, by minimising the manual handling and lifting of bricks.

3.3.2. Social aspects of recycling steel and re-using porcelain sanitary ware

Only one respondent expressed concern regarding occupational health issues in relation to the dismantling and manual transportation of sanitary ware, and claimed that the physical working environment was unsatisfactory. The concern expressed related to the manual transportation of the sanitary ware from dismantling to temporary storage, and from there to trucks for transportation. If the lifting and carrying is performed in accordance with the regulations and recommendations (Swedish National Board of Occupational Safety and Health, (1998a)), no occupational hazard is identified, i. e. the task is assessed to fall in the 'green area'.

4. Ratio-indicators of eco-efficiency

By relating the amount of CO_2-equivalents avoided per metre squared of brick wall, when using old bricks instead of new ones, to the labour time, an eco-efficiency figure of 233 kg CO_2-equivalents avoided per hour of labour is achieved (Table 1). This can be compared to corresponding numbers for other recycling or re-use activities, e.g. the recycling of steel products. For the steel recycling activities in this project the figure has been calculated to 86 kg CO_2-equivalents per hour of labour. For the re-use of porcelain sanitary ware the figure is 180 kg CO_2-equivalents per hour of labour when dismantling toilets. The figure is considerably lower for



Questions:

1. How would you describe the collaboration within the working team?
2. How would you describe your own capacity for collaboration within the group?
3. To what extent were you allowed to make your own decisions regarding the daily work?
4. To what extent could you choose tasks and methods to perform the practical work?
5. How would you describe the collaboration with teachers and instructors?
6. To what extent would you guess that this course/project has increased your chances of obtaining future employment?
7. How interested would you be in working with similar tasks in the future?
8. How important would you say that it is to increase the reuse and recycling of building components?

Fig. 2. Aspects of social sustainability in the case-study evaluation. Questions regarding psycho-social working conditions. Each question could be answered on a scale from 1 to 5, where the highest number was always the most positive.

porcelain washbasins, which weigh less, but require approximately the same dismantling time per unit as the toilets. Other figures, for instance describing energy saving or potential acidification, have been calculated correspondingly. Most of them indicate that, of the three activities studied, re-using bricks is the most efficient way to invest limited man-hours to achieve environmental benefits. The dismantling of sanitary porcelain for re-use can save more energy per hour of labour, but this is only true of toilets, and not of the lighter porcelain washbasins.

5. Conclusions regarding the case studied

The results presented above are summarised in Table 1 and Fig. 2. The conclusion regarding the sustainability of brick re-use, according to this evaluation model, is that brick cleansing and re-use is a sustainable activity from both an environmental and an economic perspective. That is, the activity can be carried out for a foreseeable time without jeopardising the environment or consuming larger economic funds than it generates. However, occupational health concerns make it impossible to ensure that the activity is sustainable from a social perspective, at least in the form that it has been organised in this project. Re-use or recycling of stainless steel products is definitely sustainable from an environmental perspective and probably from a social perspective too. The economic sustainability is substantial if re-use can be achieved, but only marginal if the products are sold for recycling. As for dismantling porcelain sanitary ware for re-use, it appears to be sustainable from all perspectives, but does not reach as high eco-efficiency values as the re-use of bricks, except regarding energy saving when re-using toilets. For all activities, it was found that the dominating cost was the cost of labour (Table 2). Transportation and energy costs have little or very little impact on the result. It was not possible to include additional costs for marketing the recycled products in this study, but it has been noted that re-used bricks are easy to sell, and sometimes fetch an even higher price than new bricks (ByggIgen, 2001). Sanitary porcelain products are also easy to sell, at least as long as the price is low in comparison to new products. The key issue seems to be the attitudes to re-used materials among constructors and consumers, as well as having an established market place for recycled goods, so customers know where to look for them. The importance of these factors has not been further analysed in this study. The Swedish National Board of Housing, Building and Planning has published studies that emphasise the importance of those factors, as well as the importance of developing quality criteria to ensure the quality of re-used construction materials (Boverket, 1998a,b).

In the demolition and construction business, time is almost always critical. The mere fact that a recycling or re-use activity is economically viable, is therefore not enough to ensure that it will be carried out. Deadlines may force companies to choose which materials to recover, and applying the results of the current study shows that: *If there is a limited resource of labour hours, or if time is short, the best sustainability strategy in this case would be to focus on the re-use of bricks, provided that the occupational health concerns can be solved.*

6. Discussion

The objective of this article was to illustrate the potential use of a model for evaluating the sustainable development of waste recycling or waste re-using activities. The point of such a model would be to guide policymakers both in municipalities and firms, to decide on an appropriate allocation of resources for optimal effects regarding sustainability. It has been shown that the suggested model can be used to perform comparisons between different activities and thereby draw conclusions about resource allocation based on the results of the evaluation.

Another advantage of using the model presented is that it ensures that a holistic view of sustainable development is maintained in the process. A local authority could use the results of such an evaluation to provide guidelines for waste treatment or for setting landfill fees for different categories of waste. Firms could use them in internal environmental guidance systems, to allocate given resources for an optimum effect. To do so, the eco-efficiency indicators, which distinguish this model from, for instance, the indicator system for sustainable production suggested by Veleva and Ellenbecker (2001), are of key importance.

For the model to work in reality, it is crucial that the sustainability assessment process itself does not require too large a work effort. One fact revealed in this case-study is that the data collection required to cover all the three aspects of sustainable development is time-consuming, and therefore costly. One reason for this is that there is a lack of up-to-date environmental performance declarations, based on life-cycle analyses of different products. The availability of such data is imperative to evaluate the environmental impact and eco-efficiency ratio-indicators. Since there are now international standards or draft standards developed both for life-cycle analyses and environmental performance declarations (Swedish Environmental Management Council, 2000), one can expect the availability of comparable data of this nature to increase in the future.

To further facilitate the data collection phase, it would be desirable to identify a smaller number of indicators for each aspect. Such key indicators would speed up the process of data collection, but might impair the quality of the sustainability evaluation. It is possible to define a smaller number of key indicators that would give a good picture of the entirety, as has been done in this study, but it is not certain that the same indicators would be the most relevant ones in another case. If wrongly chosen key indicators develop in a positive direction, while other, unmonitored indicators do not, the results might even indicate a false development towards sustainability (Mitchell, 1996). To avoid over-simplification, a certain volume of specific indicators must be upheld, and further development of the model is needed to decide the minimum number of such core indicators that would be acceptable. Such core indicators would also function as a tool to avoid the risk of a subjective or biased selection of indicators for each key aspect.

Another problem is the issue of absolute, contra relative, sustainability. A recycling activity may for instance be shown to have less impact on the environment than the production of corresponding goods from virgin raw materials, and thereby be concluded to be ‘more sustainable’ or ‘closer to sustainability’. But from an absolute perspective, the use of the product may still be causing an unacceptable impact on the natural environment. This problem could be addressed through the introduction of margin indicators, as suggested by the inventors of Sustainable Development Records (Nilsson and Bergström, 1995). However, since the real margin of different environmental aspects is seldom known or described by science, it is important to continue to work with improvements, without knowing if the margin is exceeded or not. Some margins will perhaps never be established.

Even though further development is called for, it is evident that this model for sustainable development analysis has the potential to become very useful as one tool among others when designing waste management and recycling strategies in the future.

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II

Sustainable management of combustible household waste – expanding the integrated evaluation model

Anders Klang [a,*], **Per-Åke Vikman** [a], **Helge Brattebø** [b]

[a] *Department of Natural and Environmental Sciences, Ecotechnology, Mid-Sweden University, SE-831 25 Östersund, Sweden*

[*] *(corresponding author)*
+46-63-165760 (telephone)
+46-63-165450 (fax)
anders.klang@mh.se

[b] *Industrial Ecology Programme, The Norwegian University of Science and Technology, NO-7491 Trondheim, Norway*

Abstract
A previously described model for the evaluation of sustainability in waste management has been expanded and applied to biodegradable and other combustible household waste. The model was applied to a case-study focusing on the special conditions in a municipality in the sparsely populated region of northern Sweden. In this region it is usual that the collection distances are long, the volume of waste is low and treatment facilities are remote. Four scenarios for the management of municipal household waste were compared: incineration, anaerobic digestion, composting and landfilling. A system analysis was performed to ensure that each scenario fulfil all the functions that the waste could provide (heat, electricity, fuel, and soil with a high nutrient content) and a sensitivity analysis was carried out to test the reliability of the results. The results show that the evaluation model can be used to assess the sustainability aspects of different treatment scenarios for combustible household waste. The model also allows for an individual interpretation of the results presented, depending on the choice of priorities. The effects of varying the time horizons and the difference in impact depending on what fuels are ultimately replaced in energy production are discussed.

Keywords: waste management, sustainable development, system analysis, sustainability assessment.

Introduction
During the past decades, Swedish waste management has undergone significant changes in order to achieve the overall goal of sustainable waste management (Naturvårdsverket, 2002). Some important changes are:

- The law on waste management plans – all municipalities must have an up-to-date plan for all waste that arises within its geographical boundaries. The plan must indicate how this waste will be treated, and what measures the municipality will undertake to reduce the amount, and harmfulness of the waste.
- The ordinance on extended producer responsibility for specific product categories (packaging, tyres, newsprint, electronics, cars etc)
- The landfill tax, currently approximately € 41 per metric tonne.
- The ban on landfilling of combustible waste (in effect since January 1st 2002)
- A future ban on landfilling of biodegradable waste (to be implemented by January 1st 2005)

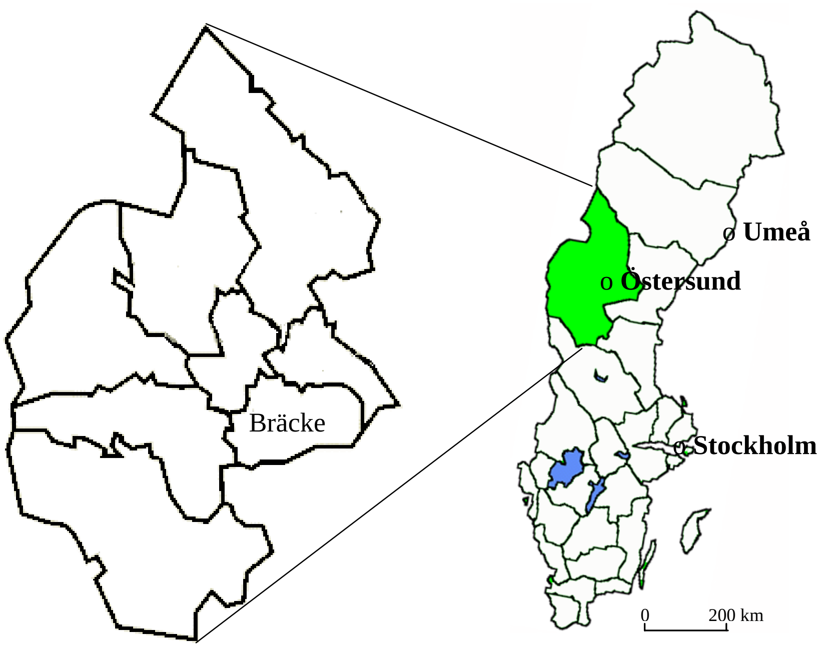

Figure 1. The municipality of Bräcke in the county of Jämtland, Sweden.

These measures have been decided after nationwide studies (Naturvårdsverket, 2002), (Naturvårdsverket, 2002), (Tillman, Baumann et al., 1991), ensuring that for the nation as a whole, they will be beneficial to the environment. However, some concern has been expressed that some of these measures are not appropriate in the sparsely populated regions in northern Sweden, due to the special conditions pertaining there. The Swedish Association of Local Authorities (2003), defines a sparsely populated municipality as:

- Low population density (<5 inhabitants per km^2)
- Few inhabitants (less than 20 000)

Other conditions of importance for waste management in such municipalities are that the collection routes are long, the volume of waste is low (typically < 2 000 metric tonnes) and treatment facilities distant (often >150 km). The municipality in this case-study, Bräcke (Figure 1) covers 3 849 km^2 and has 7 400 inhabitants, or 1.9 inhabitants/km^2.

In comparison with Swedish and European average population densities (Table 1), Bräcke can be classified as extremely sparsely populated. The land area of Bräcke consists of 78% forests, 10 % rivers and lakes, 1% farmland and 11% other types of land including built-up areas and infrastructure (Bräcke kommun, 2001). In 2002, a total of 1 047 tonnes of solid household waste was collected from households, municipal operations and industry and then transported by road to Umeå for energy recovery through incineration.

Table 1. Land area, number of inhabitants and population densities in Bräcke municipality, Sweden as a nation, and Europe.

	Land area (km²)	*Population*	*Population density (inh/km²)*
Bräcke	3 849	7 400	1.9
Sweden	410 934	8 940 788	21.7
Europe (excluding Russia)	$6{,}0 \times 10^6$	$\sim 5{,}8 \times 10^8$	~98

This case-study is part of a larger project whose main objective is to identify the essential properties and functions of an evaluation model for assessing the sustainability of waste management in sparsely populated regions. This is done through a series of case-studies previously described (Klang, Vikman et al., 2003). The present case-study focuses on the treatment of biodegradable and other combustible household waste. The objective is to examine if the integrated model suggested in (Klang, Vikman et al., 2003) can be used to assess the sustainability aspects of different waste management options for solid household wastes.

Methods

In a previously described case-study (Klang, Vikman et al., 2003), (Klang and Vikman, 2001a), (Klang and Vikman, 2001b), a model for the assessment of sustainability and the evaluation of different waste management options was suggested. The model consists of a set of indicators of three different types, and an additional set of ratio-indicators linking the former types together. This case-study uses the same model, but with modifications of the set of indicators to better match the types of waste and treatment methods analysed. The indicator categories are shown in Figure 2. The model is developed to ensure that indicators covering all relevant aspects of sustainability are considered, and can be used to provide support for decision-makers.

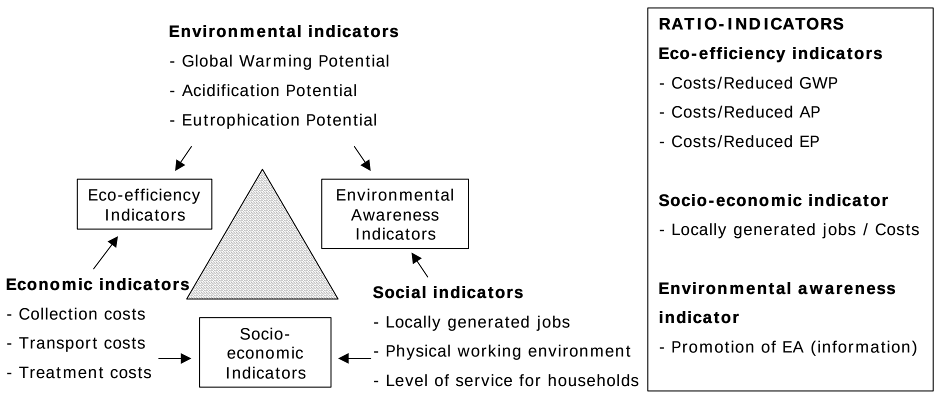

Figure 2. The categories of indicators in the suggested model

Data collection – environmental aspects

Many environmental issues could be considered in the field of waste management, but for efficient assessment an appropriate selection of the most relevant effect categories must be defined. In this case-study, much of the focus has been on emissions directly related to the treatment and transportation of waste. Greenhouse gas emissions, calculated as Global Warming Potential on a 100-year time scale, are regarded as the most important effect category to study, due to the connection between waste management and energy production systems. Since transport and combustion emissions in this region probably contribute to the eutrophication of the Baltic Sea, and the acidification of inland freshwater systems in Scandinavia, and since transportation is likely to have a larger impact on the results than in more densely populated areas (Björklund, Bjuggren et al, 2000), the data collection has also been focused on these two effect categories. For a further discussion on choice of effect categories, please refer to the paragraph Methodological considerations, in the Discussion section.

Data on the volumes of waste, collection routes and population distribution were supplied by the municipality office (Berg, 2003), (Bräcke kommun, 2001) and official statistics (Statistics Sweden, 2002). Further detailed information regarding the collection vehicles, loading capacity and fuel consumption were collected from the regional entrepreneur currently carrying out the waste collection (Hansson, 2003). Data regarding emissions and energy consumption from different treatment facilities were collected from previous research reports (Sundqvist, Baky et al., 1999), (Sonesson, 2000), (Sonesson, Björklund et al., 2000), (Leander, Rytterstedt et al., 2003). Further details of references are given in Table 2.

Data collection - economic aspects

Economic data regarding the collection and current costs for treatment and reloading, regional transport and final treatment were supplied by Bräcke Municipality (Berg, 2003), (Berg, 2003). By using a combination of the data collected in Bräcke and reports of waste system analyses (Sundqvist, Baky et al., 1999), (Sonesson, 2000), (Sonesson, Björklund et al., 2000), (Leander, Rytterstedt et al., 2003), treatment costs for alternative treatment methods were calculated. Further details of references are given in Table 2. The costs of treatment calculated include the costs of the operations, depreciation of investments and taxes. In the cost analysis, only the costs that will be paid for by Bräcke Municipality are considered. Transport and treatment costs associated with the alternative use of replaced resources are not within the system boundaries, since the objective of the model is to present consequences for Bräcke Municipality, of different waste treatment scenarios. Upstream costs associated with waste collection that are paid for by households rather than the Municipality, were examined using previous studies from (The Swedish Consumer Agency, 2003), but no monetary valuation of time spent by households on source separation has been made.

Data collection – social aspects

To obtain indicators for the physical working environment, national statistics from the Swedish Work Environment Authority were collected and analysed (Swedish Work Environment Authority and Statistics Sweden, 2003). Statistics on a more detailed level were also obtained from the same authority (Malmros, 2003). Regarding the level of service to the households, a qualitative assessment of the work effort required of the households in the different scenarios was made. An estimate of locally (within the county) generated work opportunities, has been made, as well as an estimate of the total work opportunities generated, using preliminary data on labour intensity (Klang, 2004).

Table 2. Key references for the construction of the computerised calculation model for environmental and economic indicators. References for alternative production when waste is not used are also given.

Module	*Key references*
Waste composition	(Ohlsson and Retzner, 1998)
Treatment costs	(Sundqvist, Baky et al., 1999), (Leander, Rytterstedt et al., 2003), (Östersunds kommun, 2004)
Transports costs	(Leander, Rytterstedt et al., 2003), (Statoil AB, 2002), (Berg, 2003), (Sonesson, 2000)
Waste collection	(Hansson, 2003), (Hansson, 2003), (Uppenberg, Alemark et al., 2001)
Regional waste and fuel transports	(Uppenberg, Alemark et al., 2001), (Hansson, 2003), (Björkman, 2002), (Carlsson, 2002)
Incineration	(Uppenberg, Alemark et al., 2001), (Uppenberg, Alemark et al., 2001), (STOSEB, 2001), (Zevenhoven, Skrifvars et al., 1998)
Digestion	(Sundqvist, Baky et al., 1999), (Uppenberg, Alemark et al., 2001), (de Laclos, Desbois et al., 1997)
Composting	(Smars, Beck-Friis et al., 2001), (Beck-Friis, 2001), (Beck-Friis, Smårs et al., 2003), (Komilis and Kam, 2000), (United States Environmental Protection Agency, 2002), (Mälkki and Frilander, 1997), (Tillman, Lundström et al., 1996)
Landfilling	(Ohlsson and Retzner, 1998), (Sundqvist, 1999), (Sundqvist, Baky et al., 1999)

Eco-efficiency

Eco-efficiency indicators have been defined by the following equation (World Business Council for Sustainable Development, 1999):

Eco-efficiency = product or service value / environmental influence

In this analysis a more relevant definition of eco-efficiency is to relate the difference in the costs of the waste treatment methods compared to incineration, to the differences in environmental influence achieved, measured by the reduction in the contribution to the various categories of effects previously described.

Scenario description

The analysis focused on four different scenarios, that had been considered by the municipality in question:

0. Incineration of biodegradable and other combustible household waste at the Dåva heat and power plant, in the Municipality of Umeå. Transport distance 360 km. The plant supplies district heating and electricity. Since this is the method used today, incineration is considered to be the zero-alternative to which all other scenarios are compared in the results section.
1. Anaerobic digestion of biodegradable waste in a medium-sized plant that would be situated in Östersund, 70 km from Bräcke, and incineration of remaining combustible waste in Umeå. The biogas produced to be used as fuel for heavy vehicles such as lorries and buses. Digestion residues to be composted and then used as nutrient rich soil for park and construction purposes.
2. Composting of biodegradable waste in a small-scale composting facility that would be situated in Bräcke, and incineration of remaining combustible waste in Umeå. Compost residues to be used as nutrient rich soils for park and construction purposes.
3. Landfilling of all household waste at the Gräfsåsen-plant, in the Municipality of Östersund. Transport distance 70 km. This scenario presupposes that the municipality would be granted an exemption from current and future national legislation. If such exemptions are granted, it is likely to be only for a limited time, but suggestions to seek exemption have been made from other municipalities in the region. It is assumed that the landfill used will live up to, and from some aspects exceed, forthcoming EU regulations regarding leakage control, lining and cover.

Functional unit, complementary systems, system boundaries and computational solution.

Functional unit

Different treatment methods will generate different outputs of value from the waste system, which can be used to fulfil certain functions. To enable a comparison between the scenarios, all of these functions must be fulfilled, regardless of the treatment option. This is, using life cycle assessment (LCA) terminology, referred to as the functional unit. In this case the functional unit is comprised of three different parts:

- 9,5 TJ of energy produced in a combined heat and power plant, which equals the energy that could be produced, if all the waste was incinerated.
- The equivalent of 1,2 TJ of diesel fuel for lorries and/or buses. Since methane has a lower conversion efficiency than diesel, this is the equivalent of 1,32 TJ of methane, which equals the amount of methane that could be produced, if all the biodegradable waste was digested.
- 69 tonnes of nutrient rich soil to be used for construction and municipal green surface areas, which equals the amount that would be produced if all biodegradable waste was digested or composted.

The amounts of each function has been calculated from the annual amount of waste collected in Bräcke, which was 1 047 metric tonnes in 2002, as in the zero-alternative.

Complementary systems and system boundaries

The alternative systems that fulfil the functions, when they are not fulfilled by using waste, are called complementary systems. A critical assumption with an immense influence on the results is the complementary system chosen for energy production (Gustavsson, Karjalainen et al, 2000), (Sundqvist, Finnveden et al, 2002), (Ljunggren Soderman, 2003), (Vikman, Klang et al, 2004). Which fuel is the waste replacing when it is incinerated, and what happens to this fuel? In this study it is assumed that waste will replace wood-based biofuel in Umeå, since the energy content is similar in these two types of fuel, and the use of oil in Umeå is primarily for top-load purposes. This assumption is also supported by a study investigating effects of expanded waste incineration in Sweden (Sahlin, Knutsson et al, 2004). It is also assumed that the biofuel replaced by using waste will be transported by boat to Denmark and there, in turn, replace natural gas in heat and energy production. These assumptions are based on a predicted increased demand for biofuel in an intermediate future (15 – 60 years), and that the use of natural gas is likely to increase over the coming decades. A transition to natural gas fired power plants would be a possible measure for countries to undertake in order to fulfil their commitments according to the Kyoto-protocol (Ljunggren Soderman, 2003), since natural gas plants could replace coal fired plants, which have considerably higher CO_2-emissions per unit of energy produced (Uppenberg, Alemark et al., 2001).

The complementary system for the production of vehicle fuel was considered to be traditional diesel production. Nutrient rich soil with high organic content can be produced by the extraction of peat and the addition of fertilisers.

The system boundaries for the analysis of waste treatment options and the complementary systems included are presented in Figure 3. The boundaries differ depending on which type of aspect is considered. Environmental aspects include complete complementary systems and the alternative use of the fuels that have been replaced. This means that all changes in the environmental influence from complementary systems, that are due to changes in the waste treatment systems, are allocated to the system and included in the calculations.

In the case of the assessment of the working environment, the working environment of the complementary systems is excluded. Jobs in incineration plants are assumed to be similar in all the scenarios, since the energy production is maintained with other fuels. The working environment in incineration plants is therefore excluded from the study.

Job opportunities are included if they arise within the geographical boundaries of the County of Jämtland. Costs are only included if they would be paid for by the Municipality of Bräcke. These system boundaries have been chosen since the object of the assessment tool is to provide decision support local decision-makers.

The time horizon used in the study is the intermediate future, 15 – 60 years. Calculated emissions from landfills do not include the total emissions that will occur with an infinite time horizon. Methane from landfill is assumed to be collected and incinerated without energy recovery.

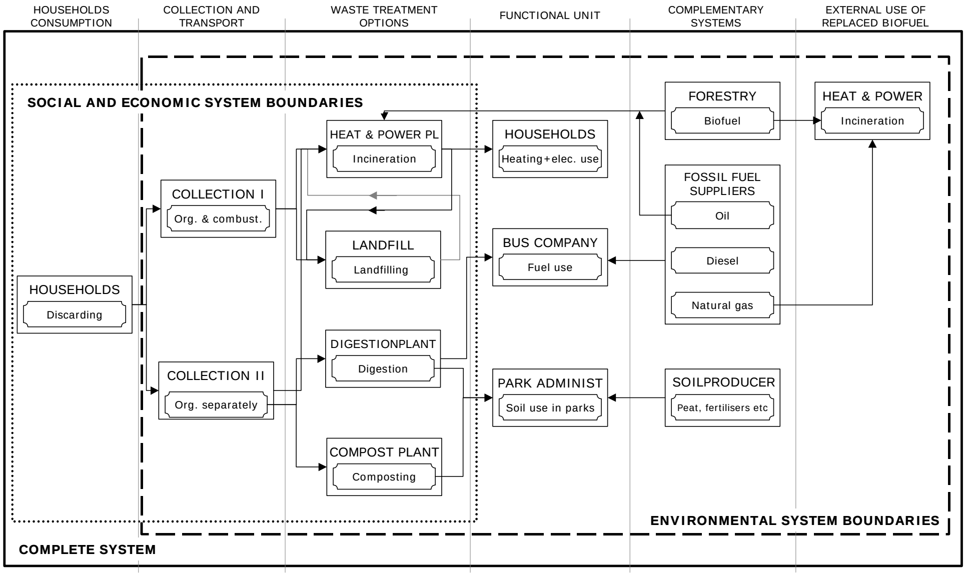

Figure 3. ***System boundaries in the study. Different aspects have been studied with different system boundaries, which is illustrated by the frames in the figure.***

Computational solution
A spreadsheet based computer model, was developed in Microsoft Excel to compute the environmental and economic aspects of different scenarios, and to calculate the results. Different spreadsheets were developed for each scenario, complete with the complementary systems. Key sources for the computation of emissions and costs are given in Table 2.

Sensitivity analysis

In the data set a number of sources with varying and sometimes unknown uncertainty have been used. Some data are of crucial importance for the results of the study, and to assess the reliability of the results, a sensitivity analysis was performed. Four variables were recognised as extra important to analyse:

1. **Energy content in waste**. Different waste incineration plants report very different energy contents in household waste (Åhgren, 1998). The reason can be a varying moisture content in the organic fraction of the waste and differences in efficiency in the separation at source of non-combustible materials such as glass and certain metal packaging etc. The default value of 10 GJ per tonne was varied by ±20% to examine the impact of this uncertainty.
2. **Treatment costs for composting and digestion.** The costs of biological treatment are uncertain, partially due to the fact that the processes are sensitive to disturbances, and this may result in additional costs. The costs per tonne of waste will also vary with the size of the treatment plant (Sundqvist, Baky et al, 1999). In the sensitivity analysis these costs were varied by ± 20%
3. **Waste volumes.** The current trend is that household waste volumes are increasing (The Swedish Association of Waste Management, 2003), but future advancement in waste prevention technologies might lead to decreasing volumes instead. The implications of ±1% annual changes in waste volumes, after separation at source of materials covered by extended producer responsibility over a 20 year period were investigated.

4. **Waste composition.** On a national level in Sweden, and also in the rest of the European Union, there is a trend away from cooking in the home towards buying ready-to-heat dishes (Nationella folkhälsokommittén, 1999). This is likely to decrease the biodegradable content of household waste over time, but without reducing the total volume, since the amount of plastic and cardboard packaging materials unsuitable for recycling will increase. The effect of a 20% decrease of the biodegradable content in the household waste was examined.

Results

The results are presented jointly for all scenarios, first from each corner stone aspect, then by ratio indicators and finally in a summarising conclusion. Since incineration is the method used today, it has been considered as the zero-alternative to which all other scenarios are compared throughout this section. The results section ends with an account of the sensitivity analysis.

Environmental results

All the treatment scenarios give higher emissions of greenhouse gases than incineration (Figure 4). This is explained by the assumption that the biofuel that is replaced by waste at the incineration plant would be transported to Denmark where it would replace fossil natural gas. Since the other scenarios require less transportation, emissions associated with boat and road transport such as $NO_{x,}$ and $SO_{x,}$, are reduced in all the scenarios compared to incineration. As landfilling takes place within the County of Jämtland it represents the minimum amount of transportation, and therefore the least emissions contributing to eutrophication and acidification (Figure 4).

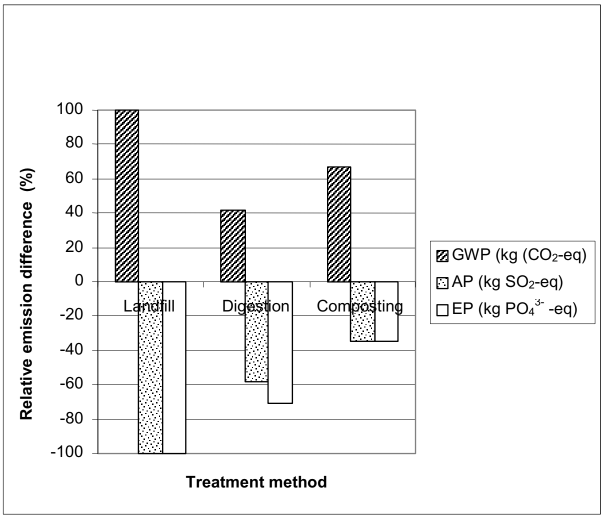

Figure 4. ***Environmental profiles of waste treatment scenarios in relation to a baseline of emissions from continued incineration. 100% equals 250x10^4 kg CO_2-eq (GWP) increase, 2,4x10^3 kg of SO_2-eq (AP) decrease and 0,29x10^3 kg of PO_4^{3-}eq (EP) decrease, compared to incineration.***

Normalisation of environmental aspects

It is difficult to assess the importance of the contribution from waste management to individual effect categories, without relating them to the community's total level of emissions. This has been done by using figures on the national emissions per capita in Sweden (Statistiska Centralbyrån, 2000), multiplied by the number of inhabitants in

Bräcke. The largest differences between the scenarios of each effect category studied (GWP, AP and EP), have been divided by the total amount of emissions in the municipality, making it possible to compare the relative size of the contribution from waste management.

Table 3. Normalisation of emissions. The largest difference in emission between the scenarios, of each effect category, is related to the total emissions from Bräcke municipality of CO_2 -eq, SO_2-eq and PO_4^{3-}

	kg CO_2-eq	*kg SO_2-eq*	*kg PO_4^{3-} -eq*
A. Largest difference between two scenarios	251 000	2 400	290
Total national average per capita emissions [a]	7 800	38	9.7
B. Bräcke totally	5.8×10^7 [b]	2.8×10^5 [c]	7.2×10^4 [d]
A/B	0.0043	0.0086	0.0040

[a] Per capita annual emissions from (Miljömålsrådet, 2003) and Swedish population Jan 1, 2003.
[b] Calculated from the annual emissions of greenhouse gases from Sweden (Miljömålsrådet, 2003).
[c] Calculated from the annual emissions of SO_2, NH_3 and NO_x from Sweden, using characterisation factors from (Lindahl, Rydh et al., 2002)
[d] Calculated from the annual release of phosphorus and nitrogen to water, NH_3 and NO_x, using characterisation factors from (Lindahl, Rydh et al., 2002) and (Swedish Environmental Management Council, 2000).

The largest difference in greenhouse gas emissions, which was found to be between landfilling and incineration, represents 0.4% of the total greenhouse gas-emissions from Bräcke Municipality (Table 3). The difference in eutrophication emissions is also roughly 0.4% of the total emissions from Bräcke, whereas acidification emissions had the largest relative importance, representing almost 0.9% of Bräcke's total emissions.

Economic results

The costs investigated under economic aspects can be divided into transport and treatment costs. The transport costs consist of two parts: waste collection and regional transport to treatment facilities. The waste collection is considered to cost the same regardless of what treatment option is chosen. This is a simplification, since the systems based on separate collection of biodegradable waste (digestion and composting) would require an investment in additional waste bins. The costs of transporting the replaced biofuel to alternative heat and power plants, in the case of waste being incinerated, are not included in the analysis, since these costs would not be paid for by Bräcke Municipality.

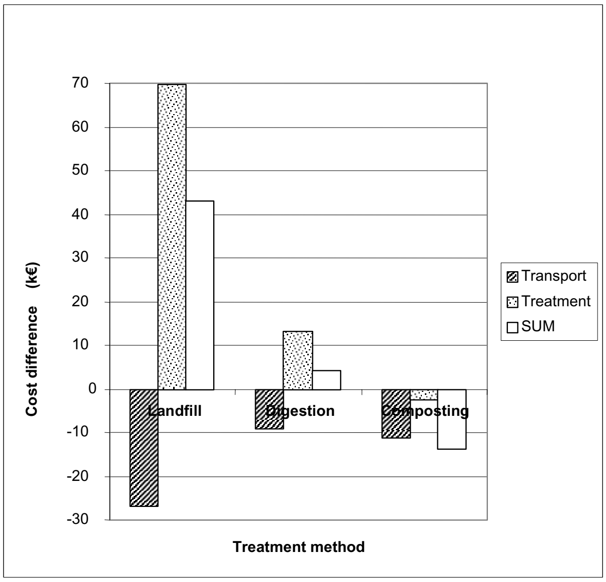

Figure 5. Annual costs for transport and treatment in waste treatment, in relation to a baseline of continued incineration. Negative bars represent lower costs than incineration, positive higher. The white bar is a summation of the other two.

Landfilling is clearly the most expensive of the scenarios, due to higher treatment costs (Figure 5). This difference is totally dependent on the current landfill tax of € 41 per tonne of waste. Without this tax, the cost of the landfill option would decrease by k€ 43 and be on the same level as the other treatment scenarios studied (Figure 6).

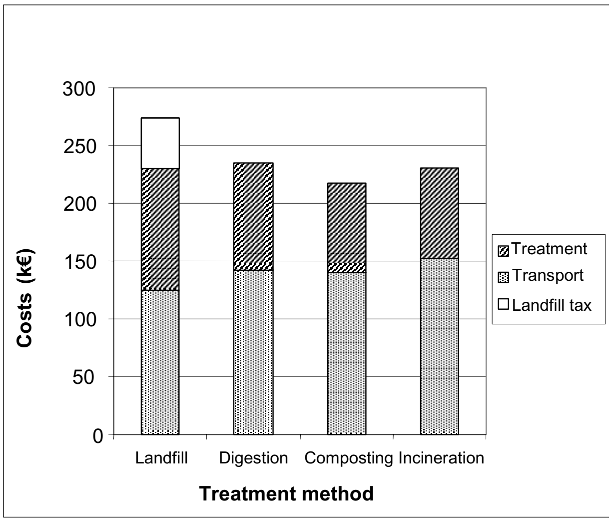

Figure 6. Annual total costs, for waste treatment. Costs shown are those paid for by Bräcke municipality for transport treatment and landfill tax. Landfill tax is also paid when waste is incinerated, but that cost is included in treatment cost.

Social results

Locally generated job opportunities

The waste collection in Bräcke municipality today employs two people working 50% each on an annual basis, or one 100% job opportunity (Hansson, 2003). The different treatment options are not considered to influence the collection phase. The collection would have to be organised in a different manner to enable the separation of biodegradable waste if the digestion or composting option is chosen, but the total work load would be similar. A difference would occur in the amount of regional transport required, and in the labour intensity of the waste treatment methods applied. The labour efforts required for transport are proportional to the transport costs.

Table 4. Locally generated job opportunities with different waste treatment scenarios.

Treatment option	*Collection*[a]	*Regional transport*[b]	*Treatment*[c]	*Sum*
Incineration	1	0.20	-	1.20
Digestion	1	0.14	0.05	1.19
Composting	1	0.13	0.08	1.21
Landfilling	1	0.04	0.10	1.14

[a] *Considered to be equal in all scenarios (Hansson, 2003)*
[b] *Calculated from total annual transport distances.*
[c] *Calculated using labour intensity indexes from (Klang, 2004)*

Thus landfilling generates the least local job opportunities in transport, and incineration the most. Thirty-one trips to Umeå per year represent one working day per week, or a 20% job opportunity, provided that there are bulk goods requiring return transport and that this work is not allocated to the waste transport. Incineration would, however, take place outside the County of Jämtland, so would not result in any locally generated jobs in the treatment process. Both composting and digestion are in some cases reported to be less labour intensive than incineration (Klang, 2004). According to labour intensity indexes expressing labour intensity per tonne of waste in different treatment facilities of different sizes described by (Klang, 2004), the biodegradable waste produced in Bräcke Municipality (356 tonnes per year) would only provide the basis for a 5% job in treatment if digested, and 8% if composted. According to the scenario descriptions, composting would take place in Bräcke Municipality, and digestion in a regional facility in Östersund. No attempt to further investigate the difference has been made. The conclusion from the employment data in Table 4 is that there is no reliable difference between incineration, composting or digestion as far as locally generated jobs are concerned, but that landfill is clearly less labour intensive.

Physical working environment

Physical working environment indicators were developed based on statistics (Swedish Work Environment Authority and Statistics Sweden, 2003), and calculated by multiplying the number of jobs generated by the risk factors for occupational diseases or accidents (Table 5). Waste collection is known to be a hazardous occupation (FEAD, 2003), and has for many years been one of the most accident and disease prone professions in Sweden (Swedish Work Environment Authority and Statistics Sweden, 2003). Since this analysis assumes no difference in the collection work, no difference is expected to occur between the different scenarios, and it is therefore excluded from the summary in Table 5.

Table 5. Physical working environment indicators for different treatment scenarios. Disease and accident risks are from (Swedish Work Environment Authority and Statistics Sweden, 2003), and represent the number of reported occupational diseases and accidents per 1000 employees in the year 2001.

Treatment	*Employees*	*Disease risk*	*Disease ind (A)*	*Accident risk*	*Accident ind (B)*	*Sum (A+B)*
Incineration						**3.9**
transport	0.2	5.8	**1.2**	13.7	**2.7**	
Digestion						**4.1**
transport	0.14	5.8	**0.8**	13.7	**1.9**	
treatment	0.05	8.8	**0.4**	20.6	**1.0**	
Composting						**4.9**
transport	0.13	5.8	**0.8**	13.7	**1.8**	
treatment	0.08	8.8	**0.7**	20.6	**1.6**	
Landfilling						**3.7**
transport	0.04	5.8	**0.2**	13.7	**0.5**	
treatment	0.1	8.8	**0.9**	20.6	**2.1**	

[a] Treatment 1 refers to the incineration of combustible, non biological material in the scenarios.

Composting is the treatment scenario most likely to cause working environment problems (Table 5), mainly because it is the method requiring the largest number of employees. Since the number of employees is very low in all the scenarios, the total risk of occupational diseases or accidents occurring is low in all cases. Statistics pertaining to the different treatment options have not been available on a sufficiently detailed level to separate the different types of waste treatment methods completely. For digestion, composting and landfilling, the same numbers have been used, and the differences are due to the difference in the calculated labour intensity.

Level of service to households

Composting and digestion require a slight increase of the work effort performed by the households, compared to landfilling and incineration. An investigation made by the Swedish Consumer Agency (Konsumentverket, 1997), concluded that the extra time required for source separation and the rinsing of packaging separated at source amounted to 18 minutes per household per week. In the present study, the increase in time due to the separation of biodegradable and other combustible waste is estimated to be roughly 10 minutes per household per week. In addition to this, the households would need to invest in equipment for the separation of biodegradable waste at source. Normal source separation bins for use under the kitchen sink cost around € 45, which can be compared to the annual fee for waste collection for an average household in Bräcke, which amounts to € 235 with a bi-weekly collection frequency (Bräcke kommun, 2002). If the bins are functional for 10 years, the extra annual cost of source separation equipment is considered to be negligible.

Ratio-indicator results

Eco-efficiency indicators

Since the environmental analysis shows that all the alternative methods release larger amounts of CO_2-eq, no eco-efficiency numbers can be calculated for reduced greenhouse gas emissions. The conclusion is that landfilling is the least eco-efficient treatment scenario, since the emission reductions obtained for the eutrophication and acidification effects are the most expensive per kg. Composting shows the highest eco-efficiency regarding these effects (Table 6).

Table 6. Eco-efficiency indicators for waste treatment scenarios, in relation to a baseline of continued incineration, measured as additional costs divided by avoided emissions, also compared to incineration. The blocked out cells indicate that no reduction of emissions is accomplished. Negative numbers indicate that the treatment method is both cheaper and contributing less to the effect category.

Costs/avoided emission	*Digestion*	*Composting*	*Landfilling*
€/CO_2-eq			
€/SO_2-eq	3	-24	18
€/PO_4^{3-} -eq	20	-142	147

Environmental awareness indicators

It has been suggested that increased separation of waste at source can be a practical method to promote environmental awareness, since the task of separating waste may lead to people thinking more about waste, and eventually changing their consumption patterns. The negative impression given by dysfunctional waste management has been shown to increase many people's environmental awareness and willingness to recycle etc (Palmer, Suggate et al, 1998), (Palmer, Suggate et al, 1999). It has been shown that there is relatively strong correlation between the willingness to source separate and the interest in buying environmentally certified products and ecologically produced food (Bennulf, 1996). If this aspect is considered, digestion and composting, which require additional source separation and therefore also more intense information campaigns to households, could also create higher environmental awareness compared to incineration and landfilling, but no attempt to quantify the magnitude of this difference has been made in this case-study.

Socio-economic indicators

Landfilling is clearly the treatment scenario that costs the most and generates the least local jobs (Table 7). The difference in the cost of treatment is of a magnitude that would influence the waste management fees for households significantly (corresponding to roughly 20% of the basic waste collection fee in Bräcke). In the other treatment methods, the difference in both costs and jobs generated is very small (Table 7).

Table 7. Socio-economic indicator describing locally generated jobs per annual cost for all waste treatment scenarios.

	Landfilling	Digestion	Composting	Incineration
Locally generated jobs[a]	1.14	1.19	1.21	1.20
Total costs (k€) [b]	274	235	217	231
Jobs/M€	4.2	5.1	5.6	5.2

[a] Job opportunities created in transport and in treatment within the county of Jämtland.
[b] Costs carried by Bräcke municipality.

Sensitivity analysis results

The input values of the four variables with the largest uncertainties were varied, and the implications of these changes were studied. On the whole, the analyses show that the results are not very sensitive to alterations in these input parameters within their assumed range of variation.

Energy content in waste
The default value 10 GJ per tonne was varied by ± 20%. This affects the functional unit of heat and energy production and in its turn, influences how much biofuel is made available to replace natural gas when waste is incinerated. The result was that the absolute size of the bars in the diagram representing the emissions of greenhouse gases varied (with ± 50 tonnes for landfilling), but the relative proportions of the different treatment methods remained unaltered.

Treatment costs for composting and digestion.
The treatment costs for composting and digestion were varied by ± 20%. This was done by following a matrix scheme, so that all possible combinations of variation were checked. When the costs of digestion were reduced by 20%, it became a cheaper solution than incineration, but it was never less expensive than composting, even if the costs of composting treatment were increased by 20%. With this combination, the difference in eco-efficiency regarding acidification and eutrophication reductions, between composting and digestion decreased substantially, but composting remained the most eco-efficient scenario. Other combinations did not influence the ranking order between the scenarios.

Volumes of waste
If the volumes of waste decreased by one percent per year as a result of improved source separation and waste minimisation strategies, the volume after 20 years would be reduced to 860 tonnes. If, instead, the current trend of increasing volumes of waste continues, and the average growth was + 1% per year, the volume would increase to 1 280 tonnes in 20 years. Both these values were tested in the model. If the volumes decreased, the differences in costs for different treatment options were reduced, but the relative proportions of the treatment types did not change. The same was true of the environmental indicators. If volumes increased over time, the differences both in economic and environmental aspects also increased, but again, the ranking order between the different scenarios did not change.

Waste composition
The effect of a 20% decrease of the biodegradable content in the household waste was studied (from 40% to 32%). This reduced the costs for the digestion option to a level very close to the costs for incineration but did not alter the ranking order between the scenarios. This reflects the fact that the cost of digestion treatment is calculated to include the depreciation costs for the investment in the plant, as well as operational costs. In reality it is only the operational costs that would be influenced by reducing the volume of organic waste, whereas the investment cost per tonne of waste would increase if the volumes were reduced. This effect is not considered in this study, since it is assumed that the waste from Bräcke Municipality would only be a minor fraction of the total waste treated in the large digestion plant.

Summary

For an overview of the results of the effects of the different scenarios studied, all the indicators are shown in Table 8. As the table is intended as a descriptive overview, no weighting was applied to the different aspects. It would be possible to use the summary to calculate a ranking between scenarios, but it would not be meaningful without a thorough discussion of the relative importance of different aspects.

Discussion

Usefulness of the model for its intended purpose

If we address the question of sustainability in waste management, rather than only looking at resource efficiency, or the economic aspects or the environmental impact, we must make a joint consideration of fundamentally different categories of effects. The model gives a framework that ensures that no relevant aspects are overlooked, but the results may appear partially contradictory or disparate. This could lead to a wish to combine all results into one summarising parameter, by using a valuation method. Normalisation could be used to assign weighting factors for prioritisation , as was done with the environmental aspects in this study. This would make it possible to compare the contribution from the waste management sector with the total contribution regarding each aspect of sustainability. In this way it would be possible to determine the relative importance of different aspects. However, as the uncertainties are too great at present for an objective weighting of different aspects of sustainability, this approach is not adopted in the present paper.

Table 8. Summarising table to illustrate the result of the various indicator analysis. The treatment methods are compared to incineration. +=>5%, ++ =>20%, and +++=>100% better than incineration, -=>5%, -- =>20% and --- = >100% worse than incineration. 0 = a difference less than 5% compared to incineration.

		Digestion	*Composting*	*Landfilling*
Environmental aspects	Greenhouse effect	---	---	---
	Acidification	++	+	++
	Eutrophication	++	+	++
Economic aspects	Transport costs	+	+	+
	Treatment costs	-	0	--
	Total costs	0	+	-
Social aspects	Generated jobs	0	0	-
	Working environment	-	--	+
	Level of service	-	-	0
Eco-efficiency	Costs/ Reduced GWP	---	---	---
	Costs/ Reduced AP	0	++	-
	Costs/ Reduced EP	+	++	-
Environmental awareness	Promotion of EA (qualitative assessment only)	+	+	0
Socio-economic	Locally generated jobs / Costs	0	+	--

As a consequence, the overview in Table 8 summarises the findings of the analysis without offering a final conclusion. In municipal waste management planning, the final decision about which treatment option to choose will be taken by the elected local political representatives. They will have varying sets of value and draw individual conclusions from the material provided by decision support tools such as this sustainability assessment. Depending on which priorities are given to different aspects of sustainability the following choices could be made:

- Priority to greenhouse gas emissions → Choose incineration
- Priority to environmental aspects and costs → Choose composting
- Priority to costs → Choose composting or incineration
- Priority to local employment and costs → Choose composting

The model allows for different priorities, but at the same time ensures that all the relevant aspects are included, and presented in a transparent manner, so that informed decisions can be made. It can also be used to highlight the need for additional effort to be made, by pointing to the critical aspects of different scenarios. In the present study this could be, for instance, that if priority was given to 'costs and employment', and the composting treatment was chosen, some additional measures should be taken to compensate for the increased emissions of greenhouse gases. The conclusion is that this application of the evaluation model has shown that it can be used successfully for the assessment of the sustainability of different waste management options for household waste, as well as for construction and demolition waste, as shown previously (Klang, Vikman et al., 2003).

Methodological considerations

Effect categories included

It is not obvious how to choose which effect categories to consider in a study of this type. It can always be argued that the results may be flawed if certain categories are omitted. Environmental effects, such as noise and toxicity will, however, be of lesser importance in this case-study than in more densely populated areas. Emissions of heavy metals depend on the content of such elements in the household waste to begin with. Different treatment methods will cause temporal and spatial differences in how the substances are released, but not alter the total emissions when studied with a sufficiently long time horizon. The main objective must in all cases be to reduce these substances in the waste from the start, through cleaner production and further improvement of hazardous waste collection services.

Working environment issues are very important to address when analysing waste management, since this sector is much more accident and disease prone than the labour market at large (Swedish Work Environment Authority and Statistics Sweden, 2003), (FEAD, 2003). The social indicators used in this study are rather crude, and basically reflect the number of job opportunities created by each waste management option. Since waste management is a high risk sector compared to the average labour market, even such crude indicators are useful, as it is likely that the individuals concerned would work in less hazardous occupations, were they not employed in waste management.

It has been suggested that this type of analysis should appoint a monetary value to the work effort spent by households. Willingness-to-pay a company to sort your waste has been suggested (Bruvoll, Halvorsen, et al, 2002), as has the cost of employing someone to sort your waste without paying general payroll taxes or social fees (Radetzki, 1999). None of these methods are generally accepted or uncontroversial, and since such costs would not influence the municipality budget the choice has been made to only assess the time required in this case-study.

A previous case-study recommended trying to limit the number of indicators studied to improve the usefulness of the model (Klang, Vikman et al., 2003). In the present paper, a manageable but relevant choice of indicators for waste management in sparsely populated regions has been made.

Avoided costs or values produced by waste treatment

In this case-study, the economic analysis has been focused on costs, rather than expected incomes or avoided costs as a result of the values produced from the waste. This aspect deserves further attention. As for the value of produced energy in the incineration plant, the income from this will not benefit Bräcke Municipality. The value of the produced nutrient rich soil can be estimated to € 970, which could be deducted from the total costs for composting and digestion, but that will only be of marginal importance.

A more important value is that of the methane produced in the digestion scenario. The value of the methane as fuel for buses can be assessed by calculating the cost of buying the corresponding amount of diesel fuel. 1.32 TJ of methane would be equivalent with 1.2 TJ of diesel, which would cost roughly k€ 21. If this value should be deducted from the total cost of digestion or not, is dependant on whether or not Bräcke Municipality would be joint owner of the digestion plant, assumed to be located in Östersund. Since this is uncertain, no such deduction has been made in this study.

Normalisation

The normalisation that was carried out for the environmental aspects indicates that changes in waste treatment methods will have a small impact on the Municipality’s total contribution to the emissions studied (Table 3). Measures focused on steps earlier in the life cycle, such as source reduction and waste preventive initiatives, hold a larger potential for improved environmental performance. However, instruments of control to promote such measures must be enforced on an international level in order to be effective (Melanen, Kautto et al., 2002), (Wilson, 1996). The normalisation results in this case might lead to the conclusion that the environmental aspects can be overlooked, and economic aspects prioritised. This conclusion should not be drawn without first making a similar normalisation of the economic results, for instance by relating the cost differences to the municipality’s total cost budget.

Uncertainties and sensitivity analysis

The sensitivity analysis shows that the results presented are reliable and robust. A large impact was seen in the greenhouse gas emissions calculated when the energy content in the waste was varied, but even then, only the size of the bars in the environmental diagrams was altered, not the relations between the different treatment scenarios. Reducing the costs of the digestion treatment by 20% made the digestion option slightly less expensive than incineration, but all in all the only really significant difference in costs is that between the landfill option and the others. This difference is completely dependent on the landfill-tax, which now seems to be at an accurate level to promote the changes it was intended for. If there had not been a landfill tax, this treatment method would have improved it’s ranking in several of the aspects studied.

A point of interest for this analysis is to examine where the job opportunities generated in complementary production would occur. The production of nutrient rich organic soil could certainly take place in the County of Jämtland, where there are a number of bogs suitable for peat extraction. No further analysis of this aspect has been made.

Regarding emissions from landfill, it has been assumed that the only type of landfill that could make possible an exemption from the bans, must fulfil and from some aspects exceed upcoming regulations regarding leakage, lining and cover. This includes a highly efficient collection and combustion system for landfill gas, and a recirculation of leakage combined with leakage treatment to prevent emissions of substances contributing to eutrophication to recipients. The effect if current practise would be used on the landfill scenarios, rather than 'best conceivable technology', has not been tested in this study.

Impacts of changes in the external energy production system

The results of a system analysis of this kind are heavily dependent on the design of the complementary system. This is true for all system analysis studies, but it becomes extremely evident when alternative fuels to energy production through incineration are involved (Finnveden, Johansson et al., 2000). When new treatment plants for waste are projected, one must consider what changes in the surrounding community can be expected to take place during the relatively long life span of the new facility. In the case presented here, one change that is likely to occur over time is in what will happen to the fuels that are replaced when waste is incinerated. If we use different time perspectives we are likely to find very different complementary scenarios. A possible development over time could be described as follows:

- **Immediate time horizon** (1-2 years). Waste would replace oil in Umeå when incinerated. If waste is used as fuel, the oil is not used elsewhere in the energy production system, since oil is currently being phased out of Swedish district heating and combined heat and power generation (Swedish Energy Agency, 2003).
- **Short time horizon** (<15 years). The alternative fuel in Umeå is wood. When waste is incinerated, surplus biofuel can be transported to Denmark by boat, to replace coal in a combined heat and power plant.
- **Medium time horizon** (15-60 years). The alternative fuel in Umeå is wood. When waste is incinerated, surplus biofuel is transported to Denmark by boat, to replace natural gas in a combined heat- and power plant (the time horizon used in this study)
- **Long time horizon** (>100 years). The alternative fuel in Umeå is wood. When waste is incinerated, less biomass will be extracted from forestry, and thus surplus biofuel will not be used.

When comparing the greenhouse gas emission results generated by varying the time horizon it becomes evident that in a distant future, when an energy system based on renewable sources can be assumed, waste incineration will not be an acceptable treatment method, at least not if the present level of plastics from fossil oil in our waste persists (Figure 7). If we consider an even longer time horizon, it is probable that the positive effect of landfilling the plastic waste in the long time horizon would be eliminated, since the carbon will probably be released eventually. On the other hand, one thing we can be sure about is that the waste of 2104 will be very different from the waste of today, so a very long time horizon is of little use to municipal decision makers.

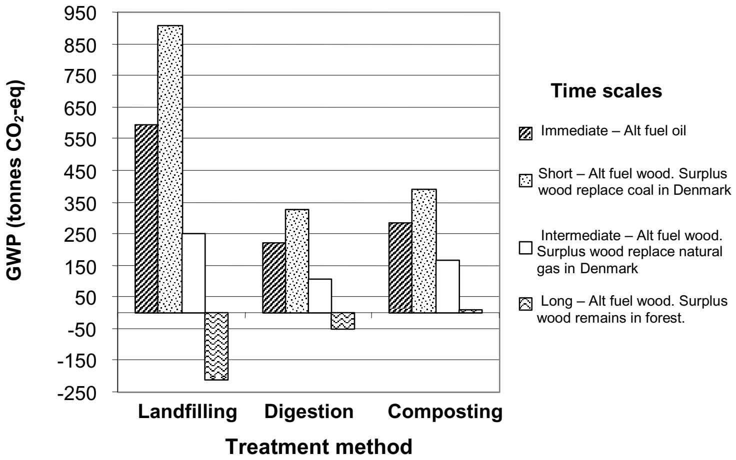

Figure 7. ***Emissions of CO_2 equivalents from different treatment methods, depending on used time horizon and assumed future development of complementary energy production systems.***

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Sustainability of wastewater treatment with microalgae in cold climate, evaluated with emergy and socio-ecological principles

Erik Grönlund[a,b,*], Anders Klang[a,c], Stefan Falk[a], Jörgen Hanæus[b]

[a] *Division of Ecotechnics, Mid Sweden University, Östersund, Sweden*
[b] *Division of Sanitary Engineering, Luleå University of Technology, Luleå, Sweden*
[c] *Industrial Ecology, The Norwegian University of Science and Technology, Trondheim, Norway*

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Abstract

The sustainability of a microalgae wastewater treatment plant model (ALGA), assumed serving a small Swedish town with 10,000 inhabitants at latitude 60°N, was tested by comparing it to a conventional three-step treatment plant (WWTP), and a mechanical and chemical treatment plant (TP) complemented with a constructed wetland (TP + CW). Using two assessment methods—the socio-ecological principles method and emergy analysis—the ALGA model considered to have a better position for sustainable development, than the other two. In emergy terms the ALGA model had about half the resource use of the other two alternatives, and used most local free environmental resources, four times the TP + CW, and 100 times the WWTP. The violations against the second and third socio-ecological principles were considered equal for the three alternatives, the fourth was estimated to be in favor of the ALGA model, and the first principle was calculated to be in favor of the ALGA model with about eight times lower indicator value sum. Recirculation of nutrients back to society or production of economically viable products from the treatment by-products would strongly influence the sustainability. The ALGA model has a potential advantage due to interesting biochemical contents in the microalgae biomass, depending on what species will become dominating.

Keywords: Sustainability; Microalgae; Wastewater treatment; Emergy; Socio-ecological principles; Cold climate

1. Introduction

The sustainability of wastewater treatment with microalgae has not, to our knowledge, yet been assessed in the scientific literature. This article is a first attempt to do so, focused on small wastewater treatment plants (serving approximately 10,000 persons) in cold areas (annual average temperature 5–6°C) and population densities in the order of 30–300 inhabitants per km^2.

1.1. Wastewater treatment with microalgae

Wastewater treatment with microalgae has since the 60s been investigated by Oswald et al. (Golueke et al., 1965; Oswald et al., 1959; Oswald and Gotaas, 1957). It was realised in a system called Advanced

* Corresponding author. Tel.: +46-63-16-5760; fax: +46-63-16-5450.
E-mail address: erik.gronlund@mh.se (E. Grönlund).

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Integrated Wastewater Ponding System (AIWPS) in full scale operation at St. Helena and Hollister in California, USA (Oswald, 1978, 1988, 1991) and in research scale from University of California, Berkeley (Green et al., 1995a,b, 1996; Oswald, 1991; Oswald et al., 1994). On research scale the high-rate pond (HRP) part of the system was described from, e.g. Israel (Shelef and Azov, 1987), Spain (García et al., 2000), France (Pagand et al., 2000), Scotland (Fallowfield et al., 1999), and New Zealand (Craggs, 2001; Craggs et al., 2000). Research on microalgae and wastewater treatment in cold climate has been performed in Canada (Kaya et al., 1995; Tang et al., 1997; Chevalier et al., 2000), though with other techniques than AIWPS or high-rate ponds. A recent more general review on phycoremediation was given by Olguin (2003).

1.2. Sustainability and sustainable development

Many methods to assess sustainability relate to the general definition of sustainable development expressed by the Brundtland Commission (1987) as: "development that meets the needs of the present without compromising the ability of future generations to meet their own needs". This general definition is often quoted, but also often criticized for being too anthropocentric and vague (Carter, 2001). Research to determine how to define the "needs" that must be fulfilled has been carried out, and often relate to work regarding basic human needs done by Manfred Max-Neef and Abraham Maslow (e.g. Håland, 1999).

Robèrt (2000) and Robèrt et al. (2002) suggested a framework for tools and concepts for sustainable development. According to this framework, five hierarchical levels can be identified, and should be distinguished when discussing sustainability. The first three levels are the principles for:

(1) The constitution of the system (e.g. ecological and social principles).
(2) A favorable outcome of planning within the system (principles of sustainability).
(3) The process to reach the favorable outcome (sustainable development).

The fourth level (4) is the action level, e.g. recycling and switching to renewable energy. The fifth level (5) is the monitoring and audit level, where, e.g. indicators are used to determine if level (4) actions are in compliance with principles for the wanted process set up on level (3), and the status of the system. On this fifth level, many efforts have been made to develop indicators to be used to monitor systems or societies (Anonymous, 1999) to see whether they are moving towards or away from a sustainable state. A sophisticated method to systemize indicators was suggested by Bossel (1999). Indicators systems focused on production systems are described by Veleva and Ellenbecker (2001), and indicators for sustainable waste management have been developed by Klang et al. (2003).

Lately, sustainability has become an important factor when discussing wastewater treatment techniques. As in many other fields there is no generally accepted method to assess sustainability of wastewater treatment, but many different approaches have been used by different authors. Sustainability of wastewater treatment has been discussed from a life cycle analysis perspective by Bengtsson et al. (1997), from an exergy perspective by Hellström and Kärrman (1997), from an emergy perspective by Björklund (2000), and Geber and Björklund (2001), and from a system analysis perspective by Chen and Beck (1997) and Hellström et al. (2000).

1.2.1. Socio-ecological principles

One method useful for directive change along a path towards a sustainable future or development is the socio-ecological principles method (Holmberg et al., 1996). The method has found widespread use in enterprises all over the world (The Natural Step, 2002), but especially in organizations and community authorities in Sweden, where it was developed. The method does not cover all aspects of sustainability, but provides the minimum requirements that are far from fulfilled in modern society (Robèrt et al., 1997). The method consists of only four principles, but these were after processing, agreed upon by a large group of scientists in a consensus document (Holmberg et al., 1996). The principles are on level (2) in the framework mentioned above (Robèrt et al., 2002).

The four socio-ecological principles are (Holmberg et al., 1996):

In a sustainable society, nature is not subject to systematically increasing:

1. concentrations of substances extracted from the earth's crust;
2. concentrations of substances produced by society;
3. degradation by physical means;
   and, in that society...
4. human needs are met world-wide.

The first principle implies that rare elements in the ecosphere often cause trouble even if taken from the lithosphere only in small amounts (Azar et al., 1996). Examples of such elements are Cd, Pb, Hg, and other heavy metals. But it also implies that large amounts from the lithosphere of substances common in the ecosphere also often cause trouble, e.g. sulphur, and fossil carbon.

The second principle points to substances produced in the technosphere, foreign to nature's degradation and recirculation system, and therefore, often persistent and accumulative in nutrient chains and loops It is acknowledged that persistent substances will accumulate in the ecosphere, as long as our society produces larger volumes than are removed through monitored technical processes. The second principle also addresses substances that are naturally existing, but for which the anthropogenic emissions threaten to disturb or disrupt natural cycles. Indicators relating anthropogenic production to natural production have been suggested as one way of monitoring this aspect of sustainability (Azar et al., 1996), where the underlying assumption is that as long as the anthropogenic production is only a fraction of the release from natural production, the risk of environmental damages as a result of anthropogenic influence is low.

The third principle recognises that it is not enough to protect the ecosphere systems from increasing amounts of disturbing substances, but the systems volume must also be maintained, to not "... reduce the physical conditions for the long-term production capacity in the ecosphere or the diversity of the biosphere" (Azar et al., 1996). Deforestation, soil erosion, species extinction, and transformation of productive land to asphalt roads are examples of violations of this principle. The total biological production and diversity must be maintained to avoid "... a loss of the productive capacity for the supply of food, raw materials and fuel. This dependence will become more obvious when the use of fossil fuels is reduced" (Azar et al., 1996).

While the first three principles concentrate on maintaining life supporting ecosphere systems, the fourth complements them, with the recognition of the problems associated with growing global population, and uneven distribution of life-supporting wealth. To meet the needs of people living today with a low standard of living, and tomorrow's increased population our use of resources must be more efficient and more equally distributed both among, and within human societies.

1.2.1.1. Socio-ecological principles as a compass for sustainable development. The four socio-ecological principles are principles of sustainability, and defines a favorable outcome towards which a sustainable development path should lead. This is visualized as the sustainable opening of a narrowing funnel (Robèrt, 2000). In this way, the principles can act as a compass for a path of sustainable development. An activity violating the principles less than other activities can be considered having a better position to enter, or continue a process of sustainable development. Consideration has to be taken to avoid blind alleys, i.e. activities that in short term go in the right direction and violate the principles to a lesser degree, but never can completely fulfil the sustainable state defined by the principles (Robèrt, 2000; Robèrt et al., 2002).

1.2.2. Emergy evaluation

Calculating resource flows in emergy evaluation differs from other methods in that the resource flows are corrected for their position in the energy hierarchy of the biosphere. The position of an item in the energy hierarchy is suggested to correspond to the relative influence of that item, on the system of which it is a part. The method has been developed the last three decades by Odum et al. (Odum, 1971, 1983, 1988, 1994, 1996; Odum et al., 2000). Wastewater treatment systems have been evaluated on emergy basis by Odum et al. (1977), Björklund et al. (2001) and Geber and Björklund (2001). In emergy accounting all inputs to the system are converted to their energy content. This actual energy content is then multiplied with a factor—called transformity—which measures the energy of one kind needed to support one actual Joule of the energy type in the system. The emergy accounting is based upon the assumption that the observations of energy flowing through hierarchical patterns in systems, mirrors a universal law, suggested by Odum (1996) as a fifth law of thermodynamics. In these hierarchies of energy or matter, units higher up in the hierarchy are assumed to have higher influence on the systems function than units lower down. This difference of influence is mirrored by multiplying the energy or matter value by a correction factor of emergy per present measurable J or kg, to receive the emergy value (with the unit solar emergy Joules, sej). If the emergy per kilogram correction factor is converted to emergy per J (sej/J) by, e.g. using Gibbs free energy, the comparable emergy per J value (the above mentioned transformity factor), describes the level in the global energy hierarchy of the geobiosphere for the item. Human work can be converted to emergy by the metabolic energy use. Differences in influence of varying works are then corrected by using different correction factors—transformities—for different types of human work. Another way of calculating human work was suggested by Odum (1984), where monetary flows were converted to emergy flows. The conversion of monetary values to emergy values are based on the assumption that the buying power of a country's GNP for a chosen year, was proportional to the total emergy use of that economy that year. The total emergy use generating the GNP for that year includes fuels, minerals, and emergy of free environmental services, both indigenous and imported. Services by humans are in this way assigned to the money paid for the service.

There is no generally accepted method how to perform sustainability assessment with emergy evaluation. Lagerberg et al. (1999) used many emergy indices to discuss sustainability of the Swedish economy, e.g. total resource use, percent indigenous renewable, percent import, and more. Many of these indices were suggested by Brown and Ulgiati (1997) and Ulgiati and Brown (1998), who also introduced an emergy sustainability index (ESI) which is a ratio between wanted yield and environmental load on the system; the ESI "... indicates if a process provides a suitable contribution to the user with a low environmental pressure". They claimed that clearly such an index "... that incorporates these aspects would shed light on sustainability issues and the fit of human economies with that of the biosphere" (Brown and Ulgiati, 1999). Odum and Odum (2001) claimed that sustainability, normally viewed as seeking a sustainable steady state, rather should be viewed as adapting to the pulsing of resources. Pulsing seems to be a general design principle for systems on all scales—"the pulsing paradigm" (Odum, 1994; Odum et al., 1995). Sustainability is then focused on finding the right strategy to sustain through the four repeating main phases of pulsing: growth, transition climax, descent, and low-energy restoration (Odum and Odum, 2001).

1.3. Objectives

This article aims to apply emergy evaluation (Odum, 1996) and a socio-ecological principles method (Holmberg et al., 1996) on a model microalgae wastewater treatment plant (ALGA) at latitude 60°N in Sweden, to view sustainability of microalgae treatment of wastewater in cold climates. For comparison, figures were presented from emergy evaluations of a conventional wastewater treatment system (WWTP) and a conventional treatment plant complemented with a constructed wetland (TP + CW).

1.4. Method

1.4.1. The comparison treatment plants (WWTP and TP + CW)

The Surahammar treatment plant (WWTP) and the Oxelösund treatment plant (TP + CW) were presented and evaluated in Björklund et al. (2001), and Geber and Björklund (2001), and key features from these evaluations are presented in Table 1. The WWTP, located 150 km west of Stockholm (annual average temperature of 5 °C, average population density of 30 inhabitants per km^2), is a conventional three-step plant, with mechanic treatment followed by simultaneous biological active sludge treatment and chemical phosphorous precipitation with iron sulphate. The sludge is treated anaerobically in a digester, and the biogas produced is used to produce electricity for the treatment plants internal use. The TP + CW, located 120 km south of Stockholm (annual average temperature of 6 °C, average population density of 300 inhabitants per km^2), use mechanical treatment (grid and aerated sand filter) followed by chemical precipitation with aluminium sulphate. The sludge is treated anaerobically in a digester. Due to higher demands on nitrogen reduction, the plant was complemented

Table 1
Design data for the Surahammar WWTP, the Oxelösund TP + CW, and the ALGA model

Plant	p.e.[a]	Mean inflow, (m^3per day)	Detention time (days)	Land area (ha)	BOD red (%)	P red (%)	N red (%)
Surahammar (WWTP)[b]	9714	5800	0.5	0.87	96	95	50
Oxelösund (TP + CW)[b]	9947	6400	9	22	95	99	50
ALGA with storage 6.5 month	10000	5800	42.5 storage 200	89	97[c]	64[c]	90[c]

[a] Person equivalents.
[b] Values from Björklund et al. (2001) and Geber and Björklund (2001).
[c] Oswald, 1991, values from St. Helena wastewater treatment plant in California, USA.

with a wetland of 22 ha, consisting of two parallel systems of shallow vegetated ponds, intermittently loaded.

1.4.2. The model (ALGA) treatment plant

The theoretical cold climate microalgae wastewater treatment plant model was designed to be comparable to the conventional wastewater treatment system of Surahammar and to the conventional treatment plant complemented with a constructed wetland (TP + CW) of Oxelösund. The ALGA model was therefore designed for 10,000 p.e. and to operate under the same climatic conditions as the Surahammar and Oxelösund wastewater treatment plants, at approximately latitude 60°N in the middle of Sweden. Since the microalgae wastewater treatment technology has a different treatment profile than the Surahammar and Oxelösund wastewater treatment plants, the ALGA model was constructed to equal influent flow and treatment performance of organic load (BOD). Björklund et al. (2001), and Geber and Björklund (2001) theoretically adjusted the Surahammar and Oxelösund plants to equal each other in nitrogen treatment performance. The ALGA model was assumed to show higher performance as to nitrogen treatment, and lower performance as to phosphorous treatment.

No published performance or design data for high-rate microalgae wastewater treatment in cold climate were found. Therefore, the ALGA model was based on experiments reviewed by Oswald (1978, 1988) and climatic data for the Surahammar–Oxelösund region of Sweden (Josefsson, 1987, 1993). The main microalgae treatment step, the high-rate pond was assumed to perform from late April to early October, which means about 5.5 months. Therefore, the first pond in the ALGA model, the facultative pond, was designed to have a storage capacity of 6.5 months. The ALGA model was assumed to produce approximately 1 vertical meter microalgae biomass sludge in the algae settling pond (ASP) per year. This means that it is not realistic to transport the sludge 7 km away as in the TP + CW case without further treatment. Therefore, a nearby microalgae biomass dewatering bed was designed in the model. A dewatering bed was also designed for the primary sludge from the facultative pond.

The design data of flows, turnover time, land area, and treatment performance are listed in Table 1 together with the Surahammar and Oxelösund, and Table 2. The model items used in the ALGA model are listed in Appendix A. A graphic view of the ALGA model is given in Fig. 1.

1.4.3. Sustainability assessment

Since there is no consensus of any operational definition of sustainability or any standard method, two different methods were chosen to view sustainability. The socio-ecological principles method (Holmberg et al., 1996) was chosen as one wide spread method claiming to be possible to use as a compass on how to change direction to a process of sustainable development according to the framework of Robèrt (2000) and Robèrt et al. (2002). The emergy evaluation method (Odum, 1996) was chosen since there were very different contributing factors important to the wastewater treatment systems, and the method claims to be able to compare these inputs of very different kinds with emergy as the common counting base. For comparison, figures were presented from emergy evaluations of the conventional wastewater treatment system of Surahammar and the conventional treatment plant complemented with a constructed wetland (TP + CW)

Table 2
Design data for the different ponds in the ALGA model

Stage	Mean inflow	Detention time	Area	Depth (m)	pond lining and maintenance
Facultative pond with storage capacity	5800 m^3 per day	20 days 200 days	29 ha (430 × 670 m)	4	Unlined but graded pond bottoms
Primary sludge dewatering bed	11000 m^3 per year	1 year	2.68 ha (40 × 670 m)	0.4	Walls of excavated masses, 5 cm gravel in bottom, freeze-dried sludge removed once a year
High-rate microalgae pond (HRP)	13050 m^3 per day	6 days	26.1 ha (416 × 630 m)	0.3	16 m wide straight channels unlined but graded pond bottoms, lining of concrete around paddlewheels, on site soil dividers of 2 m width
Algae settling pond (ASP)	13050 m^3 per day	1.5 days	0.66 ha (33 × 200 m)	3	Lined and graded pond bottoms, plastic lining, settled microalgae biomass removed once a year[a] to Algae dewatering bed
Algae biomass de-watering bed (ADB)	6600 m^3 per year	1 year	1.66 ha (83 × 200 m)	0.4	Walls of excavated masses, 5 cm gravel in bottom, freeze-dried sludge removed once a year
Maturation pond (MP)	13050 m^3 per day	15 days	19.6 ha (311 × 630 m)	1	Unlined but graded pond bottoms

[a] The settled microalgae biomass in the ASP need to be removed in intervals of about 6–12 months (Craggs, 2001), which in this model means once a year.

of Oxelösund evaluated by Björklund et al. (2001), and Geber and Björklund (2001). The socio-ecological principles method was also applied on these two treatment plants.

1.4.3.1. System borders. Both the emergy evaluation and the socio-ecological principles methods use a systems perspective. Systems overview (or analysis) commonly views input and outputs of energy, available

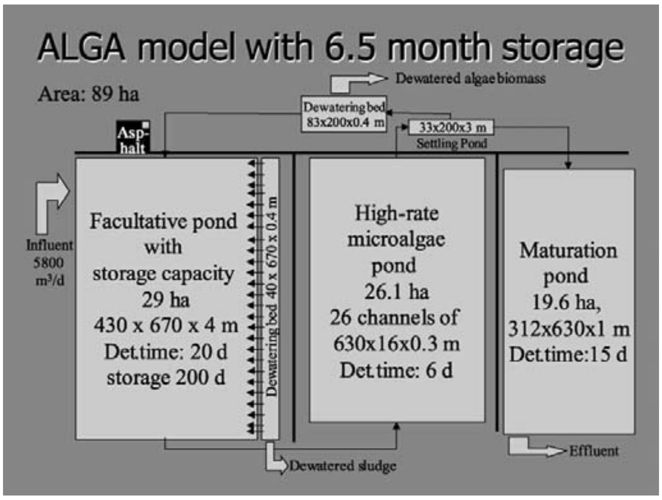

Fig. 1. The ALGA model—graphic view of area relations.

energy (exergy), material or money, to and from a defined system. Sometimes also information in- and outputs are considered, as well as different scales in space and time (Ulgiati, 2001). In our systems overview we listed inputs of energy, material and money to the microalgae wastewater treatment model, as were done by Björklund et al. (2001) and Geber and Björklund (2001) for the Surahammar and Oxelösund wastewater treatment systems. The system border in space was set to the wastewater treatment plant. The piping to the plant was omitted from the calculations, as were the recipient. The system border in time were set to a treatment plant life length of 50 years reallocated to steady state flows on a one year basis, i.e. the initially high construction flows of energy, material and money were divided by the estimated life length of the constructions, to represent annual flows.

Both purchased inputs and "free" environmental inputs to the treatment systems were considered in the emergy evaluation. The free inputs were treated as the same source of geobiospheric work and therefore not possible to sum up (see footnote e in Table 3). Only purchased inputs were used in the socio-ecological principles assessment.The quality of the data used varied: the data for the Surahammar plant (WWTP) and the Oxelösund plant (TP + CW) were based mostly on measured data from Björklund et al. (2001) and Geber and Björklund (2001), and for the microalgae wastewater treatment model (ALGA) mostly calculated data were used. The construction costs were reestimated as built 1996 for the WWTP and the TP + CW, to be comparable to the ALGA model.

1.4.3.2. The socio-ecological principles method. The socio-ecological principles were used and interpreted as follows:

Azar et al. (1996) suggest indicators of lithospheric extraction rates (human extraction per natural weathering and volcanic activity), and accumulated lithospheric extraction (accumulation in the technosphere compared to ecospheric content), as possible indicators to estimate status of the first principle. For the sustainability analysis in this case, indicators of the former type were used in a quantitative method to assess which of the three suggested treatment systems violates the first principle the most. This was done by recalculating resource use down to use of ground elements, and multiplying the total weight of each element with the indicator value given by Azar et al. (1996) for that element. For complex materials, consisting of more than one element, indicator values were calculated based on content of different elements. By adding the resulting numbers, a crude quantitative assessment of the violations against the first principle was made, allowing a comparison between the different systems.

Violations of the second principle, regarding increasing concentrations of anthropogenic substances in the environment were addressed with a qualitative approach. A special focus was set on the use of chemicals in the waste water treatment, and on emissions of substances from the treatment plants.

To assess a system's performance in accordance with the third principle, the total constructed area demanded was used, as suggested by Azar et al. (1996). That is land covered by concrete, asphalt or other non biologically productive surfaces.

Violations of the fourth principle were evaluated in a qualitative/quantitative way by comparing the different systems use of available free resources (flowing water and incoming sunlight, etc.) as opposed to use of human made, unevenly distributed resources.

The alternative violating the four principles least was considered as having a better position to enter, or continue, a process of sustainable development leading to sustainability according to Robèrt (2000) and Robèrt et al. (2002). No weighting process, to compare violations against different principles to one another, was performed. The concept of flexible platforms (Robèrt, 2000; Robèrt et al., 2002) to avoid activities that in short term appear more favourable, but are unlikely to lead to sustainability, was not considered in this investigation.

1.4.3.3. The emergy evaluation method. The emergy accountings were performed according to Odum (1996). However, the evaluation table was modified to also fit data for the socio-ecological principles method. To estimate the emergy of human work, the emergy to dollar ratio for Sweden, 1996 (Lagerberg et al., 1999) was used, following the suggestion that the emergy and monetary flows are proportional (Odum, 1984). It was assumed that none of the monetary flows were biased by special monetary valuation following fluctuations on a short time scale, e.g. fashion. The emergy to dollar ratio for Sweden 1996

Table 3
Systems overview of the resource use in energy, material, monetary, and emergy terms for the Surahammar WWTP, the Oxelösund TP+CW, and the ALGA model

Item	Energy, mass or money per year			Emergy per unit sej/(J, kg, USD)	Solar emergy E+15[a] (sej per year)		
	WWTP*	TP + CW*	ALGA		WWTP*	TP + CW*	ALGA
Free envirnoment inputs							
Solar insolation[b] (GJ)	18	460000	2530000				
Wind[c] (GJ)	44.7	3000	1630				
Rain[d] (GJ)	26.9	685	2750				
Geobiospheric work[e] (GJ)	26.9	685	2750	1.82E+04[a]	0.49	12.5	50.0
Solar insolation[f] (GJ)	18	460000	2530000	1	0.00	0.46	2.5
Wind[f] (GJ)	44.7	3000	1630	1500	0.07	4.5	24.4
Rain (GJ)	26.9	685	2750	1.82E+04	0.49	12.5	50.0
Land cycle[f] (GJ)	Not calculated	Not calculated	Not calculated		Not calculated	Not calculated	Not calculated
Collected seedlings[g] (kg)		215 1.34 GJ		2.11E+03[a]		0.0	
Collected seed[h] (kg)		72 0.06 GJ		1.06E+04[a]		0.0	
Loss of topsoil[i] (kg)		16000 kg 1.70 GJ		7.40E+04[a]		0.1	
Total free energy, material, and emergy (kg)	90 GJ	464000 GJ 16500 kg	2534000 GJ		0.5	12.6	50.0
Purchased inputs							
Electricity[j] (GJ)	2513	3960	1270	1.19E+05[a]	299	471	151
Oil[k] (GJ)	1020	47.8	30	6.60E+04[a]	67.6	3.15	2.0
Iron—total[l] (kg)	20900	180	60	2.65E+12[a]	55.4	0.47	0.16
Machinery[m] (kg)	1140	40	320	4.10E+12[a]	4.67	0.16	1.3
Copper[n] (kg)	125	12.5	1.25	6.80E+13[a]	8.50	0.85	0.08
Chemical precipitation[o]–							
$FeSO_4$ (kg)	320000						
Fe^{2+} (kg)	24000			2.65E+12[a]	63.6		
Al SO_4 (kg)		429650					
Bauxite (kg)		129000		1.50E+10[a]		1.9	
Electricity (GJ)		60.3		1.19E+05[a]		7.2	
Concrete[p]	103000 kg	118000 kg	7800 kg	7.34E+11[a]	75.6	86.6	5.7
Bricks[q]	1670 kg	1780 kg		2.52E+12[a]	4.21	4.48	
Gravel[r]	0	0	61000 kg	5.00E+11[a]			30.5
Asphalt[s]	34500 kg	5800 kg	65500 kg	4.74E+11[a]	16.4	2.75	31.0
Rock wool[t]	325 kg	71.3 kg	2.8 kg	1.84E+12[a]	0.60	0.13	0.01
Plastic[u]	20 kg	147 kg	430 kg	5.87E+12[a]	0.12	0.86	2.5
Polymer[v]		2000 kg		See appendix		3.0	
Wood[w]			17 kg 0.35 GJ	6.6E+03[a]			0.0
Purchased inputs—service[x]							
Investment[z] cost (kUSD)	120[y]	110[y]	80	2.14E+12[a]	258	236	171
Interest[A] (kUSD)	300[y]	275[y]	200	2.14E+12[a]	645	591	428
Operation and maintenance cost[B] (kUSD)	291	463	37	2.14E+12[a]	625	995	79
Cost of land[C] (kUSD)	0	3	12	2.14E+12[a]	0.00	6.45	26
Total energy, material, money and emergy of purchased resources	3533 GJ 482000 (kg) 710[y] kUSD	4008 GJ 558000 (kg) 850[y] kUSD	1300 GJ 135000 (kg) 330 kUSD		2123	2411	928
Total resource use for wastewater treatment	3600 (GJ) 482000 (GJ) 710[y] (kUSD)	468000 (GJ) 574000 (GJ) 850[y] (kUSD)	2535000 (GJ) 135000 (GJ) 330 (kUSD)		2123	2424	978
Energy and emergy in the influent wastewater*	15900 (GJ)	15900 (GJ)	15900 (GJ)	3.76E+06[a]	59790	59790	59790

Table 3 (*Footnote continued*)

* WWTP, TP+CW, and the influent wastewater values were adapted from Björklund et al. (2001) and Geber and Björklund (2001), and the calculations behind these values are given in those papers and commented below only if recalculated in some way in this paper. For full calculations please contact the corresponding author.

[a] E+04 = $\times 10^4$, E+NN = $\times 10^{NN}$.

[b] WWTP: solar energy not obviously used in treatment process. TP + CW: solar energy not obviously used in TP process, but in wetland. Sunlight used in photosynthesis, and for heating of wetland. But probably the wetlands would work as nitrogen treatment step also if theoretically covered. 220000 m^2 wetland, of which 130000 are vegetated with albedo 0.12, and 90000 are non-vegetated with albedo 0.20, solar irradiation 67726 cal cm^{-2}, gives 460 TJ (Geber and Björklund, 2001). ALGA: solar energy used in ponds in photosynthesis and for heating of ponds. Facultative pond and high-rate pond relies on microalgal photosynthesis, and solar energy works as disinfection in maturation pond. Algae settling pond does not use solar energy, and would work theoretically also if covered. The dewatering beds use sunlight for drying, and the 10 ha of land area use the solar energy but are not obviously used in the treatment process. Calculations, see footnote e below

[c] WWTP: wind energy not obviously used in treatment process. TP + CW: wind mixing energy used in wetland. ALGA: wind mixing energy used in ponds wind drying energy used in dewatering beds. Calculations, see footnote e below.

[d] TP + CW: rain energy probably used by plants in covered part of wetland. ALGA: the rain energy are used by the 10 ha of land area, but do not obviously contribute to the treatment process. Calculations, see footnote e below.

[e] In the geobiosphere work are included the inputs from sun, tide energy and deep earth heat, giving rise to the complex pattern of streams in oceans, winds and rain in the atmosphere and the geologic land cycle. To determine in the local analysis how much of the geobiospheric work should be allocated to the system of attention, there is a risk of double counting if different energies are added. It is a complicated task to subtract out the double counting part from each input. Odum (1996) suggests a simpler way to determine how much solar emergy the earth system has contributed by using the largest of the geobiospheric inputs and ignore the rest as double counting. In all three evaluated wastewater treatment plants rain contributed most emergy of the geobiospheric inputs (land cycle are considered lowest of the inputs and therefore not calculated) and therefore will represent the combined work of sun, wind, rain, and land cycle. *Solar insolation*: item already included in rain, footnote d. Transformity 1 sej J^{-1} per definition (Odum, 1996). 892000 m^2, Albedo 0.20 in open water areas, solar irradiation 67726 cal cm^{-2}, gives 2.53E+15 J $year^{-1}$. *Wind*: item already included in rain, footnote d. Transformity 1500 sej J^{-1} (Odum, 1996). 1.83E+07 J m^{-2}(Geber and Björklund, 2001), 892000 m^2, gives 16.3E+12 J. *Rain*: chemical potential transformity in rain 18.2E+3 sej J^{-1} (Odum, 1996). Precipitation 0.624 m (Björklund et al., 2001), 892000 m^2, Gibbs free energy in rain water 4.94 J g^{-1}, gives a chemical potential energy in rain of 2.75E+12 J.

[f] Excluded from total geobiospheric work to avoid double counting, see footnote e.

[g] TP + CW: 860 specimen = 215 kg = 1.34 GJ (Geber and Björklund, 2001).

[h] TP+CW: 72 kg collected seeds = 0.06 GJ (Geber and Björklund, 2001).

[i] TP + CW: 65 m^3 soil, 0.25 mg dry matter m^{-3} = 1.70 GJ (Geber and Björklund, 2001).

[j] Transformity 1.19E+5 sej J^{-1} (Björklund et al., 2001). Pumping between dams: 233 MWh for seven pumps (equals 200 MWh for six pumps in CW (Geber and Björklund, 2001)), equals 8.39E+11 J $year^{-1}$. Driving paddlewheel: Energy of mixing ponds 7–50 kWh/ha day (Oswald, 1988) 160 days per year, 26.1 ha of high-rate pond, gives a range of 4.21 ± 3.30E+11 J $year^{-1}$. Heating of O&M house: 10 m^2, 0.2 MWh m^{-2}, gives 2 MWh or 7.20E+09 J $year^{-1}$. Total = 839 + 421 + 7 = 1270 GJ $year^{-1}$.

[k] Refined oil transformity 6.60E+4 sej J^{-1} (Odum, 1996). Sludge removal from the primary sludge dewatering bed every year, approximately 30 m^3 day^{-1}, 365 days, gives 11000 m^3 primary sludge production per year. The sludge from the facultative pond is assumed to be freeze-dried (Hellström and Kvarnström, 1996) in 0.4 m thick layer on a 40 m wide and 670 m long shelf with walls of excavated masses along one side of the facultative pond to achieve about the same dry matter as in CW. Volume assumed to decrease from 11000 m^3 of 1% dry matter to 24% dry matter (same as Geber and Björklund, 2001), which gives approximately 460 m^3. In the CW 1283 m^3 (308 ton) of 24% dry matter were transported with 4.78E+10 J. Maximum load 10 m^3, 14 km average distance, 10 km h^{-1} tractor speed, 7 L diesel per hour, 37.8 MJ dm^{-3} diesel (Geber and Björklund, 2001)). A 460 m^3 gives 46 trips, which gives 451 L diesel per year or 17 GJ diesel per year. Sludge removal from Algae dewatering bed every year: estimated to 1 m depth of settled algae biomass in algae settling pond, gives 6600 m^3 of approximately 1% dry matter. Freeze-dried to achieve 24% dry matter gives a volume of 275 m^3 algae biomass sludge per year. The transport needs for this is 28 trips, gives 275 L diesel per year or 10.4 GJ diesel per year. Fuel for construction (human work for fuel drilling and refining are included in the construction cost): two bulldozers à 20 days work, 50 L diesel per day for each bulldozer, 50 years life length, gives 40 L diesel per year or 1.5 GJ diesel per year. Total: 17 + 10.4 + 1.5 = approx. 30 GJ diesel per year.

[l] Pig iron transformity 2.65E+9 sej g^{-1} (Buranakarn, 1998). Iron reinforcement in concrete is approximately 35 kg m^{-3} in floor and wall, (Björklund et al., 2001). Iron sheet in roof, 1.25 mm thick, weight of 10 kg m^{-2}. O&M house, reinforcement in concrete floor: 10 m^2, 0.3 m thickness, 50 years life length, gives 2.1 kg per year. O&M house roof: 20 m^2, 50 years life length, gives 4 kg per year. High-rate pond concrete reinforcement, 672 m^2, 0.1 m thick, 50 years life length, gives 47 kg per year. Facultative pond removal grit: 50 kg iron, 50 years life length, gives 1 kg per year. Total = 2.1 + 4 + 47 + 1 = approximately 60 kg $year^{-1}$.

[m] Estimated as only steel. Steel end-products transformity 4.10E+9 sej g^{-1} (Brown et al., 1995). Pumps: seven á 100 kg, 20 years life length, gives 35 kg per year. Paddlewheel motor: 50 kg, 20 years life length, gives 2.5 kg per year. Machinery for construction: 2 bulldozers à 2 tonnes, 1/12 of annual machinery use (1 month of work), 20 years life length, gives 17 kg per year. Sludge transport: 1 tractor and a 10 m^3 trailer of total 3 tonnes, life length 20 years, 19 days use of total 221 annual working days, used every year, gives 260 kg per year. Total = 35 + 2.5 + 17 + 260 = 320 kg per year.

[n] In electric cables, rough estimate 1% of WWTP = 1.25 kg $year^{-1}$.

[o] WWTP: ferrous sulphuric acid 320 Mg, ferrous content (Fe^{2+}) 24 Mg per year (Björklund et al., 2001). TP + CW: 429650 kg per year of ALG ($AlSO_4$). Bauxite use per tonne ALG manufactured = 300 kg Mg^{-1}, gives 120000 kg bauxite per year. Energy use of electricity, per tonne ALG manufactured = 39 kWh Mg^{-1}, gives 60.3 GJ electricity per year (Geber and Björklund, 2001).

Table 3 (*Footnote continued*)

[p] Transformity 7.34E+8 sej g^{-1}, density, 2260 kg m^{-3} (Björklund et al., 2001). WWTP: Buildings 3130/50 Mg per year = 62.6 Mg per year. Sludge storage 248/50 Mg per year = 5.0 Mg per year. Pipes in WWTP 381.5 Mg, 50 years life length = 7.63 Mg per year. Maintenance: 3% of concrete in pipes 381.5 Mg = 11.4 Mg per year. Additional concrete for *N* treatment = 16.4 Mg per year (Geber and Björklund, 2001). Total 103 Mg per year. ALGA: O&M house, concrete floor, 10 m^2, 0.3 m thickness, 50 years life length = 136 kg per year. High-rate pond bottom around paddlewheels, 12 paddlewheels, 6 m width, 4 m of 0.1 m thick concrete lining before and after, 0.5 m vertical sides, 50 years life lenght, gives 3.04 Mg per year. Maintenance: 3% of lining in high-rate pond, gives 4.6 Mg per year. Total = 0.136 + 3.1 + 4.6 = approximately 7.8 Mg per year.

[q] No bricks in the ALGA, but can of course be used.

[r] Transformity 5.00E+09 sej g^{-1} (Lagerberg et al., 1999), density of rock approximately 2.8 tonnes m^{-3}, pack volume of gravel assumed to approximately 50%. Primary sludge dewatering bed: 5 cm layer, 2.68 ha, 50 years life length, gives 27 m^3 per year. Algae biomass dewatering bed: 5 cm layer, 1.66 ha, 50 years life length, gives 17 m^3 per year. Total = 27 + 17 = 44 m^3 per year = 61000 kg per year.

[s] Transformity 4.74E+8 sej g^{-1}, 100 kg asphalt m^{-2} (Björklund et al., 2001). O&M roads in the area, 1300 m + 2 × 700 m, 3 m wide, 5000 m^2 at O&M house, 50 years life length, gives 26.2 Mg per year. Maintenance: 3% of asphalt, gives 39.3 Mg per year. Total = 26.2 + 39.3 = 65.5 Mg per year.

[t] Transformity 1.84E+9 sej g^{-1}, density 3.50E+4 g m^{-3}, 50 years lifetime (Björklund et al., 2001). A 10 m^2 O&M building, about 40 m^2 insulated area, gives 2.8 kg per year.

[u] Polyvinyl chloride transformity 5.87E+9 sej g^{-1} (Buranakarn, 1998). PVC pipe, density 3.5 kg m^{-1}, 50 years life length (Geber and Björklund, 2001). Plastic pipes Inlet to facultative pond 6 m, primary sludge bed to facultative pond 40 pipes á 1.5 m, gives 60 m, Facultative pond to high-rate pond 350 m, high-rate pond to algae settling pond 350 m, algae settling pond to maturation pond 6 m, maturation pond to recipient 500 m, drainage in Algae dewatering bed 3 m × 83 m, algae dewatering bed to facultative pond 350 m =1900 m, gives 130 kg per year. Plastic lining PVC 0.5 mm thick, 50 years life length, in Algae settling pond, bottom 6600 m^2, 3 m depth, two parallel channels 200 m long with end sides of 16.5 m, gives: 9198 m^2, which gives 85 kg per year. A 12 paddlewheels in glassfiber, 6 m long, 1.8 m diameter, six paddle blades of 0.5 cm thickness, two end sides of 1.8 m diameter and 1 cm thickness, density of glassfiber assumed to 800 kg/m^3, gives approximately 220 kg per paddlewheel. The 20 years life length, gives 132 kg per year. A 3% maintenance per year, gives 80 kg per year. Total = 130 + 85 + 132 + 80 = approximately 430 kg per year.

[v] TP + CW: Electricity, gas and oil in production of polymer: (1.57E+10 J electricity with transformity 1.19E+05 sej J^{-1}) + (1.70E+10 J gas with transformity 4.80E+04 sej/j) + (4.09E+10 J oil with transformity 6.60E+05 sej J^{-1}) = 3.0E+15 sej per year (Geber and Björklund, 2001)).

[w] Transformity of wood products: 6.6E+3 sej J^{-1} (Lagerberg et al., 1999). A 10 m^2 O&M building, life length 50 years, wood density 300 kg/m^3, gives approximately 17 kg per year, 2.052E+7 J/kg wood products (Lagerberg et al., 1999) gives: 0.35 GJ per year.

[x] Monetary figures for service are given in approximately 1996 SEK. The emergy per money ratio for Sweden 1996 were 2.15E+11 sej/SEK (Lagerberg et al., 1999). If WWTP and TP + CW are considered to be representative for wastewater treatment in Sweden approximately 2000E+15 sej per 10000 persons, gives 0.2E+15 sej per capita for wastewater treatment. Total resource use per capita for Sweden 1996 was estimated to 40.7E+15 sej per capita. The wastewater treatment sector therefor used approx. 0.5% of the total annual emergy flow in Sweden 1996. A corrected energy per money ratio for the wastewater treatment sector would be (values from Lagerberg et al. (1999): ((total resource use 1996) − (wastewater treatment sector resource use))/(gross domestic product 1996) = 2.14E+11 sej/SEK. Ten SEK = approximately 1 USD.

[y] Recalculated value in this investigation.

[z] Emergy per money ratio for the wastewater treatment sector in Sweden 1996 were 2.14E+11 sej/SEK, see above footnote x. Construction cost are from 2002, but price index was only about 4% lower in 1996, which will not significantly change the figures (Swedish Statistics, http://www.scb.se/snabb/priser/Kpi80sv.asp 18 February 2002). WWTP*:* If the WWTP was built today (2002) the construction cost would probably be around 60 MSEK (pers.comm. Conny Simonsson, Vatten Östersund 11 March 2002). Fifty years life length gives 1.2 MSEK per year. TP + CW: If the TP + CW was built today the construction cost would probably be 50–60 MSEK (pers. comm. Jan Friberg, VAI VA-projekt AB 11 March 2002). 50 years life length gives 1.0–1.2 MSEK per year. ALGA: if the WWTP was built today (2002) the construction cost would probably be around 30–50 MSEK (pers.comm. Conny Simonsson, Vatten Östersund 11 March 2002). Fifty years life length gives 0.6–1.0 MSEK per year = approximately 0.8 MSEK per year. Ten SEK = approximately 1 USD.

[A] Emergy per money ratio for the wastewater treatment sector in Sweden 1996 were 2.14E+11 sej/SEK, see above footnote x. Interest estimated as in (Geber and Björklund, 2001), where half the construction cost is multiplied with 10% interest rate, and 50 years lifetime, gives: WWTP: annual interest: (60E+6)(1/2)(10%) = 3.0 MSEK per year. TP + CW: annual interest: (55 ± 5E+6)(1/2)(10%) = 2.5–3.0 MSEK per year. ALGA: annual interest: (40 ± 10E+6)(1/2)(10%) = 1.5–2.5 MSEK. Ten SEK = approximately 1 USD.

[B] Emergy per money ratio for the wastewater treatment sector in Sweden 1996 were 2.14E+11 sej/SEK, see above footnote x. O&M labour*:* one person 8 h per month (2 h per week) during storage period. One person 42 h per month (2 h per day) during high-rate pond period. Two persons 3 days for harvesting/desludging Algae settling pond every autumn. Two persons 3 days for harvesting/desludging the facultative pond every autumn. One person 19 days to transport the freeze-dried primary sludge and algae biomass to landfill, if four trips per day (footnote k). This gives approximately 3.2 working month per year = approximately 130 kSEK per year. O&M administration: About one working month per year = about 40 kSEK per year. O&M purchased energy: electricity; price (Björklund et al., 2001): 106 E−09 SEK/J, gives 135 kSEK. Oil: price (Björklund et al., 2001) = 94 E-09 SEK/J, gives 2.80 kSEK. O&M purchased material: concrete 3% per year, 4.6 Mg per year, 1/2.26 m^3 Mg^{-1}, 1000 SEK m^{-3} (http://www.angelfire.com/nc/ballastnord/pris.html 2002-03-11), gives 2.0 kSEK. Asphalt 3% per year, 393 m^2 per year, 162 SEK m^{-2} (Kjell Jonsson, 2002. pers.com. Vatten Östersund, Östersunds kommun), gives 64 kSEK. Plastic pipes and machinery neglectible. Total = 0.13 + 0.04 + 0.138 + 0.064 = 0.37 MSEK per year. Ten SEK = approximately 1 USD

[C] Emergy per money ratio for the wastewater treatment sector in Sweden 1996 were 2.14E+11 sej/SEK, see above footnote x. Pond and dewatering bed area: 79.36 ha. Asphalt area: 1.31 ha. O&M house area: 0.0010 ha. Ten meters space between and around main dams (minus asphalt area): 4.9 ha. Two meters dividers between high-rate pond raceways: 25 × 2 × 630 = 3.15 ha. Algae settling pond divides in two parts for sludge handling, 2 m between: 2 × 200 = 0.04 ha. Maturation pond divided in four parts for flow management, 2 m between: 3 × 2 × 630 = 0.38 ha. Total = 79.36 + 1.31 + 0.001 + 4.9 + 3.15 + 0.04 + 0.38 = 89.2 ha. Same cost as CW: (30000/22 SEK ha^{-1} per year = 1360 SEK ha^{-1} per year), (89.2 ha)(1360 SEK ha^{-1} $year^{-1}$) = 122000 SEK per year. Ten SEK = approximately 1 USD.

(Lagerberg et al., 1999) was corrected for double counting of the emergy allocated to the wastewater treatment sector of the Swedish economy, in accordance with Odum (1984). The total resource use, and two indices—percent local renewable and an emergy sustainability index according to Brown and Ulgiati (1997), and Ulgiati and Brown (1998)—were used to view sustainability. The pulsing view on sustainability of Odum and Odum (2001) was not considered.

The alternative showing the least resource use, the highest part of renewable resources, and the highest emergy sustainability index, was considered as having a better position to enter or continue a process of sustainable development according to Robèrt (2000), and Robèrt et al. (2002).

Odum (2000) suggests an improvement of the baseline for emergy accountings. The baseline is the total annual emergy inflow to the biosphere from the sun, moon, and deep heat sources. The baseline emergy is the driving force for everything physically happening in the biosphere, together with storage driving forces (e.g. fossil fuels, ores, forest biomass, soil organic content) previously built up by the annual emergy inflow. In this evaluation we used the older baseline (Odum, 1996) since we related to Björklund et al. (2001) and Geber and Björklund (2001) who used the older baseline. To convert the values in this investigation to the new baseline, increase the emergy values by a factor of 1.68. To convert values the other way to compare with this investigation decrease by a factor 0.60.

2. Results

2.1. Systems overview

The energy inputs on an annual basis (Table 3), were totally dominated by solar insolation in the TP + CW and the ALGA model. The energy inputs in the WWTP were dominated by purchased electricity. Of the purchased inputs in the TP + CW and ALGA model electricity was the main energy source. The annual material inputs (Table 3) were dominated by chemicals for precipitation in the WWTP and TP + CW, and concrete was also a major input in these systems. The dominating material input in the ALGA model was asphalt and gravel, with concrete also as a significant input. The annual monetary inputs (Table 3) were dominated by the capital investment and the interest for all three plants, with the operation and maintenance costs also dominating in the WWTP and the TP + CW. The land costs were almost neglectible, however much higher for the ALGA model than the WWTP, with the TP + CW in between.

Treated wastewater to a certain level was the common output from all three treatment plants, as were sludge, however of different quality: the sludge from the WWTP and TP + CW were precipitated iron respectively aluminum phosphate sludge, the sludge from the ALGA model was primary sludge from the facultative pond and algae biomass sludge from the algae settling ponds. An output from the TP + CW was the wetland as habitat for water fowl and other organisms, and as a recreation area for people using the paths and small operation and maintenance roads around the wetland. This is a possible output also in the ALGA model, at least for the maturation pond.

2.2. Socio-ecological principles method results

2.2.1. First principle

The quantitative assessment of the first principle showed that the first principle—about not systematically increasing concentrations of substances extracted from the earth's crust in the biosphere—was violated by all the treatment alternatives, but most by the WWTP and the TP + CW and least by the ALGA model, see Table 4. The use of oil, chemical precipitation and asphalt have the largest impact on the result.

2.2.2. Second principle

For the second principle—about not systematically increasing concentrations of substances produced by society in the biosphere—the following qualitative observations were made: the contents of unwanted and toxic substances in the wastewater were more a question of influent wastewater quality, which was considered to be the same in all three alternatives and not within the system borders for our analysis. Methane from the constructed wetland (CW) in the TP + CW adds to the greenhouse effect. The chemicals used for precipitation in the WWTP and TP + CW are considered to be relatively rapidly degradable and are not known to have long-term harmful effects on the productivity of the ecological systems.

Table 4
Substances violating the 1st condition. Indicator values (dimensionless) from Azar, Holmberg et al. (1996)

Footnote[a]	Substance	Raw unit per year			Recalculated by indicator values			
		WWTP (A)	TP+CW (B)	ALGA (C)	Indicator values	Ind × WWTP	Ind × (TP + CW)	Ind × ALGA
j	Electricity (1/3 of total)[b] (GJ)	840	1320	420	0.0027	2.3	3.6	1.1
k	Oil[c] (GJ)	1020	47.8	30	130.3	132907	6228	3909
l	Iron – total (kg)	20900	180	60	1.4	29260	252	84
m	Machinery[d] (kg)	1140	40	320	1.4	1596	56	448
n	Copper (kg)	125	12.5	1.25	24	3000	300	30
o	Chemical precipitation[e] (kg)	320000	430000		0.26/0.60	84592	257992	
p	Concrete[f] (kg)	108000	118000	8810	0.053	5776	6311	471
q	Bricks[g] (kg)	1670	1780		0.057	96.1	102.4	
r	Gravel[i] (kg)			61000	0.057			3510
s	Asphalt[h] (kg)	34500	5800	65500	0.39	13393	2252	25427
t	Rock wool[f] (kg)	325	71.3	2.8	0.057	18.7	4.1	0.16
u	Plastic[h] (kg)	20	147	430	5.2	103.8	762.8	2232
v	Polymer[h] (kg)		2000		5.2		10378	
Sum of indicator values × weight						270000	280000	36000

[a] Footnotes see Table 3.
[b] Recalculated as kilograms of extracted uranium, corresponding to the used electricity (Vattenfall, 2001).
[c] Recalculated as kilograms of carbon, using carbon content in heating oil (Swedish Environmental Protection Agency, 2002).
[d] Calculated as iron.
[e] Calculated from content of iron and sulphur (WWTP), and aluminium and sulphur (TP + CW) (Kemira Kemwater, 2001).
[f] Calculated from a mixture of cement and the chemical composition of medium rock from Swedish lithosphere (Swedish University of Agricultural Sciences, 2002).
[g] Calculated as Swedish medium rock (see above).
[h] Calculated to consist of 6% bitumen (oil-based) and 94% Swedish medium rock (above).
[i] Calculated as carbon, using medium carbon content of common combustible plastics (Greenpeace Sweden, 2000).

Based on this qualitative reasoning it was considered that there were no clear differences between the treatment alternatives as far as the second principle is concerned.

2.2.3. Third principle

For the third principle—about not systematically increasing degradation by physical means in the biosphere—the following observations were made: on one hand the WWTP uses the least area, the ALGA model the most. On the other hand both the TP + CW and ALGA model switches productive areas to biological production of another kind. All treatment plants use biological activity in the process, but of different kinds: WWTP—bacterial activated sludge and anaerobic stabilisation, TP + CW—wetland biological production, ALGA model—bacterial activity and algal production. The biological activity (measured as produced biomass) may even be increased by replacing original productive area by the WWTP. This may also be the case for the TP + CW and ALGAs as pointed out in the outputs in the systems overview. Apart from the physical core of photosynthetic productive and resource producing areas one could argue that the ponds of an ALGA model provide open space, possible recreational areas, parklands incorporating the ponds and habitat for wildlife, particularly water fowl. The same can be said for the constructed wetland in the TP + CW solution. Another approach is to calculate the size of the built area required by the different systems. That is, the area covered by buildings or roads (concrete and asphalt), which will take the place of natural vegetation and not allow any biological production. All three solutions required slightly less than 1 ha of constructions (not including the ponds), thus from this respect no difference between them was seen. All together it was considered that there was again no clear difference between the alternatives.

Table 5
Aggregated emergy flows and emergy indices

Flow or index	WWTP	TP + CW	ALGA
R, local renewable, items 4–6 in Table 3 (sej per year)	0.5E+15	12.5E+15	50.0E+15
N, local non-renewable, item 7 in Table 3 (sej per year)		0.1E+15	
F, purchased resources from society, items 8–25 in Table 3 (sej per year)	2123E+15	2411E+15	928E+15
U = R + N + F, total resource use (sej per year)	2123E+15	2424E+15	978E+15
EYR = U/F, emergy yield ratio	2123/2123 = 1.000	2424/2411 = 1.005	978/928 = 1.054
ELR = (F + N)/R, environmental load ratio	2123/0.5 = 4246	(2411 + 0.1)/12.5 = 193	928/50.0 = 186
SI = EYR/ELR, sustainability index	0.24E−3[a]	5.2E−3[a]	57E−3[a]
Percent local renewable = R/U (%)	0.02	0.5	5.1

[a] E−3 = $\times 10^{-3}$.

2.2.4. Fourth principle

The fouth principle—about that human needs must be met world-wide in a sustainable society and that resource use is efficient and fair—was violated by all treatment alternatives. From the perspective of equity in resource use, one could argue that the WWTP uses more of unfairly distributed, partially imported materials such as oil, iron and copper (Table 3) while the other two use more free local resources. The TP + CW plant requires use of chemical precipitation and a higher use of bought energy in the form of electricity and oil than the ALGA model. The conclusion is that the ALGA model was the most resource efficient solution except when considering land use, and that the ALGA model violates the fourth principle the least.

2.2.5. All four principles

The ALGA model violated the four socio-ecological principles less than the other two treatment systems, since it was favorable from the perspectives of the first and fourth principles. As for the WWTP and TP + CW, the socio-ecological principle analysis cannot say which of those systems were violating the principles least.

2.3. Emergy evaluation results

The emergy flows in the different wastewater treatment alternatives are listed in Table 3, with calculations in the footnotes of Table 3, and summarised in Table 5.

Table 5 shows that the ALGA model used less than half of total (U) and purchased (F) resources compared to the TP + CW and the WWTP. The ALGA model used most local renewable emergy, the WWTP least, and the TP + CW in between. The calculated sustainability index for the ALGA model was approximately 10 times higher than the TP + CW, and more than 200 times higher than the WWTP. The percent local renewable followed the same pattern.

The dominating input in all three treatment systems were monetary inputs of service and electricity. Significant inputs were also in the ALGA model gravel, asphalt, and geobiospheric work, in the TP + CW concrete, and in the WWTP oil, iron, and chemicals for precipitation, and concrete. Other inputs were very small in emergy terms.

3. Discussion

3.1. Systems overview

The large energy inputs from insolation yielded very rough figures, as did the estimations of wind and rain energy. It was difficult to assess how much solar and wind energy that was actually needed for the function of the constructed wetland and the ALGA model ponds. Photosynthesis, heating, and disinfection were the obvious functions for solar energy and circulation for wind energy. The sun and the wind also provide evapotranspiration for the above surface vegetation in the CW. The chemical energy in rain was not obviously used in the WWTP and the ALGA model; the ALGA model would probably show the same function with a plastic roof channelling the rainwater away. In the CW it may be used by the vegetation above water surface. They were however unquestionable energy flows crossing the system border, and it is not clear how the system would work without them. Due to this and the fact that they were used in the emergy evaluation (Geobiospheric work in Table 3), we put these energy flows in our resource use overview, though rough estimations.

The domination of chemicals for precipitation in the WWTP and TP + CW material input were not surprising since precipitation of phosphorous is the main goal in Swedish wastewater treatment. It is interesting that the constructed wetland built to meet higher demands on nitrogen removal does not increase the material inputs very much. The dominating material input of asphalt in the ALGA model could be considered not needed at all for the function of the plant, but would probably be built in a real situation, to improve the maintenance.

The economic inputs were the most difficult ones to estimate for the ALGA model. They are probably slightly lower in construction cost, and much lower in operation and maintenance cost compared to the WWTP and TP + CW, but the actual levels are difficult to estimate. A better estimate would need actual cases to rely on.

The different outputs of the treatment alternatives must of course be considered. The sludge from all three treatment alteratives should be able to use for biogas production, and the iron and aluminium phosphate sludge can be used as fertilizer and soil improver (Kvarnström, 2001) if it is not to much contaminated with unwanted substances. The ALGA model sludge also has potential for use as fertilizer and soil improver (Metting et al., 1990), but the most interesting outcome from the ALGA model is the quality of the microalgae biomass sludge, that depending on species composition have higher potential of producing economic viable products than chemical precipitation sludge (Grönlund et al., 2001).

Systems overviews of input–output type are generally not good in taking into account area properties (O'Neill, 2000). In this case it is very important to point out the very different land demands of the treatment systems. The ALGA has a very large area demand, and the WWTP a very low.

3.2. Socio-ecological principles

The socio-ecological principles method showed that the violation against the second and third principles were considered equal for the three compared alternatives, the fourth were estimated to be in favor of the ALGA model, and the first principle was calculated to be in favor of the ALGA model with about eight times lower indicator value sum (Table 4). Which one of the WWTP and TP + CW that violated the principles least could not be distinguished in this investigation.

The socio-ecological principles high-lighted the higher use of renewable resources in the ALGA model and TP + CW, compared to the WWTP. The high electricity need, and also in some cases, material inputs in the TP + CW, may be partially explained by the fact that the plant probably is oversized, as discussed in Geber and Björklund (2001) (designed for 19,000 person equivalents, personal communication Jan Friberg, VAI VA-projekt AB 11 March 2002).

There is a problem in comparing different material inputs to each other as was done to evaluate violations against the first principle. The simplified method of multiplying weights of different elements to an indicator, describing the relation between human extraction and natural weathering and volcanic activity releasing that element, does not result in a complete understanding of impacts on a life-cycle basis. For instance, large extractions of materials that are not known to cause any environmental problems, might be considered more important than small extractions of substances that have large environmental impact, or where the extraction itself causes considerable damage. This is illustrated by the case of uranium extracted for electricity production, which has a very small impact on the result of the analysis. But it is well known that extraction of uranium causes considerable damage to natural ecosystems and the use of uranium to produce electricity results in difficult waste management problems. Those aspects of uranium mining should on the other hand be covered by the second, third and fourth principles instead. In this analysis such aspects were omitted.

The fact that human carbon extraction is 6.4 times higher than the "natural extraction" (Azar et al., 1996) has a large influence on the result through the use of oil, asphalt, and plastics. The crudeness of the method does not allow any separation of the WWTP and TP

+ CW solutions, but the difference compared to the ALGA model system is large enough to conclude that the latter violates the first principle the least.

In the discussion about release of fossil carbon dioxide from materials, algae biomass can of course be used as a carbon sink, if the biomass is not oxidised which would release the carbon dioxide again. Such methods could also be applied on organic sludge from WWTP and TP + CW, and peat production in the CW. But such use of the biomass would not allow a nutrient recycling to arable land, which in turn would serve as an incitement for continued extraction of phosphorus from the lithosphere, thereby causing an further violation of both the first and fourth principles.

In the context of global warming potential, concrete should also be considered a carbon dioxide source, since carbon dioxide is released when lime is burned. This carbon dioxide is fixed again when the concrete is weathering (the carbonisation process). The time span for this is, however, in the vicinity of 200 years (Börjesson and Gustavsson, 2000), which certainly is a key period for the environmental problem of the greenhouse effect. In our investigation the carbon dioxide emissions from lime burning was not considered in the results.

3.3. Emergy evaluation

In the emergy evaluation the ALGA model was considered having a better position for sustainable development compared to the two other alternatives, since it used less than half of the total and purchased emergy (Table 5). The ALGA model also used most local free resources, more than the TP + CW, and both of them much more than the WWTP, because of the larger area of the ALGA model and the CW.

The calculated sustainability index was very low for all three wastewater treatment systems (Table 5). Within this low level, the sustainability index for the ALGA model was more than 200 times higher than the WWTP, and approximately 10 times higher than the TP + CW. The percent local renewable index followed the same pattern. This is what to expect since there were more free environmental energy used in the TP + CW and the ALGA model, and the environmental load (defined as purchased and local non-renewable emergy flow per local renewable emergy flow) was much lower mainly due to the larger area. From an empower density view (emergy per time and area) the sustainability index for wastewater treatment processes will show low values as long as it need to treat wastewater flows from high empower density areas as cities are today in our fossil fuel based society. In a future eventually based on less fossil fuels the empower following the wastewater from urban settlements would be lower, and the sustainability index for wastewater treatment processes may increase.

The main driving force for the three wastewater treatment systems was unfortunately also the one coupled to most uncertainty in the ALGA model: the roughly estimated monetary inputs were by far the largest input. One way of interpreting this fact is that the sustainability of the wastewater treatment plants depends largely on the sustainability (resource use, emergy use) in the surrounding society, more than the actual material and energy use in the treatment plant. The high emergy flow allocated to interest can be interpreted as a “permission” to use more actual emergy in the initial investment phase.

Not included in this evaluation was if the microalgae biomass could be used in some way in society. This could lead to a lower raw resource input to the society. If the microalgae biomass contains economically valuable biochemicals, the cost of the wastewater treatment would decrease, as it also would if methane is produced from the sludge or in AIWPS fermentation cells (Green et al., 1995a), and could replace other purchased energy. These improvements would however require inputs of devices for extraction of biochemicals or collection of methane. Ødegaard (1995) claimed that recovered methane would improve sustainability in wastewater treatment. Björklund et al. (2001), however, found no support for this in emergy terms. If an economically viable product could be produced from the microalgae biomass, it would outcompete the other methods both from a traditional market-economy perspective as well as a sustainability perspective. This said with reservation for dense populated regions where land costs can be to high for this large area demanding technology.

3.4. The microalgae wastewater treatment plant model

The ALGA model turned out to use much more land area than the other two alternatives. The size of the

ALGA model is however depending on what treatment is in focus. If nitrogen reduction is the main focus, and the goal is 50% (as it is at the coastal wastewater treatment plants in the Swedish part of the Baltic region), the ALGA model could be downsized considerably (N-reduction 90% lowered to 50%, see Table 1). To meet the from a Swedish perspective low phosphorous treatment performance, the ALGA model could be complemented with chemical precipitation as in the fellingdams of northern Sweden (Hanaeus, 1991; Pettersson, 1997). This would of course also change the material inputs. One very important note on the ALGA model is that the model was constructed without experimental data from the actual or even similar regions. To be more reliable it needs support from future experimental data. The suggested treatment performance, Table 1, was inferred from the AIWPS plant in St. Helena, California. This plant, however, has fermentation cells, which decrease the sludge production and probably increase the disinfection security level (the total emergy use in the ALGA model may decrease approximately 3–4%, if fermentation cells were introduced, due to less sludge production). It is not possible to estimate the fermentation cell part of the BOD, nitrogen and phosphorous treatment performance data. Other investigations of high-rate ponds give a supporting picture—even though not clear—of treatment performance regarding BOD, nitrogen and phosphorous assumed here.

To overcome problems with too low algae production in wintertime storage capacity in the facultative pond was chosen in this article. Other possible solutions are the use of artificial light, e.g. fluorescent light tubes and/or covering and heating of the high-rate pond. This alternative was considered to expensive by De la Noüe et al. (1992).

3.5. Sensitivity analysis for the ALGA model

The largest uncertainty was the construction cost of the ALGA model. The estimated annual uncertainty of ±20,000 USD with corresponding interest of ±50,000 USD, was approximately 16% of the total emergy use. The uncertainty of the electricity use for driving paddlewheels was about 4% of the total emergy. The sensitivity of other flows were much lower than this, giving a total uncertainty in the order of ±20% of the emergy use. This does not affect our conclusions. The ALGA model used much more land than the other two alternatives. A land cost in the order of 5–6000 USD per ha and year would double the emergy use of the ALGA model, bringing it to the same order as the WWTP and TP + CW.

4. Conclusion

From a sustainability view the theoretical microalgae wastewater treatment plant (ALGA model) was considered to have a better position to enter or continue a process of sustainable development as defined by Robèrt (2000), and Robèrt et al. (2002), than the conventional three-step treatment plant in Surahammar (WWTP), and a conventional mechanical and chemical treatment complemented with a constructed wetland in Oxelösund (TP + CW), when sustainability was evaluated with the socio-ecological principles method (Holmberg et al., 1996) and with emergy evaluation (Odum, 1996, and Brown and Ulgiati, 1999).

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Appendix A. Model items

For purchased energy and material, and economic input in the ALGA model. Generally assumed 50 years life length, further information on calculations, see footnotes (fn.) in Table 3.

A.1. Capital costs

- Investment cost, fn. (z).
- Interest, fn. (A).
- Operation and maintenance cost, fn. (B).
- Cost of land, fn. (C).

A.2. Facultative pond

- Pond excavation, 29 ha and 4 m depth, fn. (k, m).
- Influent pump, 20 years life length, fn. (j, m).
- Grit removal, fn. (l).
- Plastic pipes, inlet to facultative pond, fn. (u).
- Sludge pump to dewatering bed, fn. (j, m).
- Plastic pipes: facultative pond to high-rate pond, fn. (u).
- Pump, Facultative pond to High-rate pond, 20 years life length, fn. (j, m).

A.3. Primary sludge dewatering bed

- Walls of excavated masses, fn. (k, m).
- Bed bottom, 40 m × 670 m, fn. (r).
- Plastic pipes from gravel bed to facultative pond, fn. (u).
- Sludge transport to landfill every year, fn. (m).

A.4. High-rate pond

- Pond excavation, 26 channels of 630 m × 16 m and 0.3 m depth, fn. (k, m).
- 12 paddlewheels (estimated from http://www.earth-rise.com 5 february 2002), fiber glass, 6 m long, 1.8 m in diameter (Oswald, 1988), 20 years life length, fn. (u).
- Paddlewheel hydraulic motor activated by a 3 hp electric motor (Oswald, 1988), fn. (j, m).
- Concrete bottom around paddlewheels, 4 m before and after, fn. (p).
- 3% yearly maintenance of concrete, fn. (p).
- Iron reinforcement in concrete, fn. (l).
- Plastic pipes, high-rate pond to algae settling pond, fn. (u).
- Pump, high-rate pond to algae settling pond, 20 years life length, fn. (j, m).

A.5. Algae settling pond

- Pond excavation, two channels of 200 m × 16.5 m and 3 m depth, fn. (k, m).
- Plastic pipes, algae settling pond to maturation pond, fn. (u).
- Plastic lining PVC 0.5 mm thick, fn. (u).
- Pump, algae settling pond to algae dewatering bed, 20 years life length, fn. (j, m).
- Pump, algae settling pond to maturation pond, 20 years life length, fn. (j, m).

A.6. Algae biomass dewatering bed

- Walls of excavated masses, fn. (k, m).
- Bed bottom, 83 m × 200 m, fn. (r).
- Sludge transportation, fn. (k, m).
- Plastic pipes: drainage in algae dewatering bed, and from algae dewatering bed to Facultative pond, fn. (u).

A.7. Maturation pond

- Pond excavation, four channels of 630 m × 78 m and 1 m depth, fn. (k, m).
- Plastic pipes: maturation pond to recipient, fn. (u).
- Pump, maturation pond to recipient, 20 years life length, fn. (j, m).

A.8. O&M house 10 m^2

- Concrete in floor, fn. (p).
- Iron reinforcement in concrete floor, fn. (l).
- Roof, sheet iron, fn. (l).
- Wood panel in walls, fn. (v).
- Rockwool insulation, fn. (t).
- Heating with electricity, fn. (j).

A.9. Infrastructure

- Asphalt roads in the area, fn. (s).
- Asphalt area around O&M house, fn. (s).
- 3% yearly maintenance of asphalt, fn. (s).

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IV

Systems analysis as support for decision-making towards sustainable municipal waste management – a case study

Anders Klang [a,*]***, Per-Åke Vikman*** [a]***, Helge Brattebø*** [b]

[a] *Department of Natural and Environmental Sciences, Ecotechnology,*
Mid-Sweden University, SE-831 25 Östersund, Sweden

[*] *(corresponding author)*
+46-63-165760 (telephone)
+46-63-165450 (fax)
anders.klang@mh.se

[b] *Industrial Ecology Programme, The Norwegian University of Science and Technology,*
NO-7491 Trondheim, Norway

ABSTRACT: During the last decade, systems analysis has become a more frequently used tool in municipal waste management. This paper investigates how one such analysis, carried out in a Swedish county, was perceived by local municipal officers and politicians, as support in the decision-making process. A questionnaire was sent out to municipal officers and local politicians in local government committees and municipal councils. Among the most important aspects in evaluating scenarios, the respondents emphasised possibilities for municipal co-operation to minimise cost and negative environmental influence, sound working conditions for refuse disposal personnel, low emissions of greenhouse gases, keeping household economy in mind and that suggested technologies are known and reliable. Aspects of relatively low importance were the number of locally generated job opportunities and minimising the work efforts for the households. The study also showed differences between male and female respondents and between politicians and municipal officers, on how scenarios were valued, and on which aspects of the system-analysis were of greatest importance for this valuation. Respondents, on average, were satisfied with the system-analysis, and its usefulness as a decision-support tool. However, more work should be carried out to explain and present the results of the systems analysis to further improve its usefulness.

Keywords: systems analysis; waste management; decision-making; waste treatment, municipality

Introduction

Systems analysis in Waste Management

During the last decade, decision support tools based on systems analysis have been developed around the world. Theoretically based on general systems theory, and systems engineering practices, they are mostly used for decision support in relation to activities that will demand large investments in infrastructure, where it is important that the chosen technology will be considered appropriate for a long period of time. The energy sector (Vikman *et al.*, 2004), (Finnveden *et al.*, 2003), (Schlamadinger *et al.*, 1997) and the waste management sector (Oostra, 1996), (Finnveden, 1999), (Swedish Environmental Research Institute, 2002), (Dewulf *et al.*, 2002), (Klang *et al.*, 2003), (Eriksson, 2003) are two examples of areas where systems analysis has become increasingly common. The intuitive objective of a systems analysis for waste management might be to give a picture as complete as possible of the consequences of different technological options, often aiming at describing both environmental, economic, and sometimes also social effects. Thus, a systems analysis will give the decision-makers the opportunity to make an informed, and hopefully more sustainable decision. In order to facilitate such an analysis it is often necessary to develop and/or apply computer-based models for waste management and different treatment options (Barlishen K. D. *et al.*, 1996). In Sweden, systems analysis tools for waste management have been developed in close co-operation with active research groups, most notably the ORWARE and MIMES Waste groups, (Sundberg, 1993), (Bjorklund *et al.*, 1999), (Sundqvist *et al.*, 1999), (Ljunggren, 1997). Lately regular consultant firms have also started to take on such analysis commissions (Leander *et al.*, 2003). Some work has been done to evaluate the quality of such systems analysis in terms of reliability and relevance (Leander, 2002), but so far not much has been done to investigate how they function in the decision processes of the municipalities, or what can be done to improve their usefulness (Eriksson *et al.*, 2003). This case is based on a systems analysis for waste management in the county of Jämtland, Sweden, which was performed in a project where the municipalities of Jämtland have co-operated with the private firm Carl Bro AB (Leander *et al.*, 2003). The objective of this case study is to investigate which aspects elected representatives and municipal officers regards as the most important ones to include in a waste management systems analysis, and how they perceive the value and usefulness of a systems analysis as a decision support tool.

The county of Jämtland

Jämtland is a sparsely populated county in the northern part of Sweden. The county covers 49 400 km^2, and has less than 130 000 inhabitants divided on eight municipalities. Almost half the population lives in the municipality of Östersund, in the county centre (Figure 1).

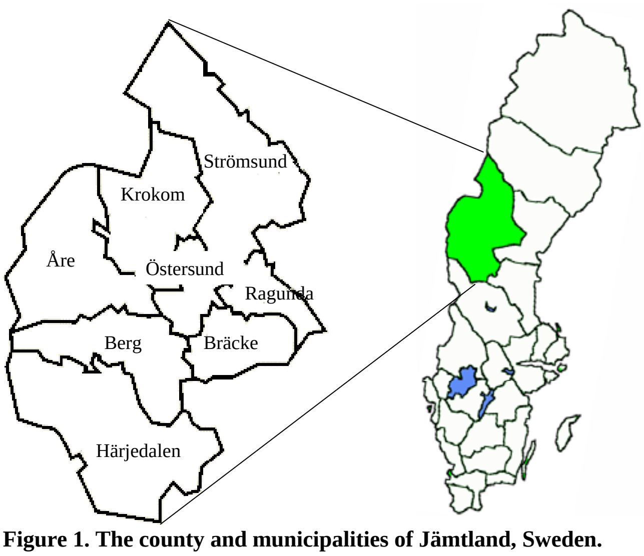

Figure 1. The county and municipalities of Jämtland, Sweden.

From a waste management point of view, this makes the region special, since collection and transportation work will have a higher impact on the results of a systems analysis than what is often the case (Björklund *et al.*, 2000). The systems analysis under study, can shortly be described as an investigation of four different possible scenarios for future waste management in the region, with environmental and economic implications of each scenario described. The main reason for performing the systems analysis was the upcoming ban on landfilling of biodegradable waste that will take effect on January 1, 2005. The group of municipal officers and political decision makers that commissioned the analysis decided that it should only investigate scenarios that were technically and economically feasible today, and at the same time adequate to fulfil current and expected future national goals and regulations for waste management. In addition to economic and environmental consequences the analysis also contain results regarding social aspects in terms of demands on public participation and awareness and created job opportunities as well as technical aspects such as systems flexibility and reliability of different waste treatment technologies. The four studied scenarios were:

0. **The zero-alternative.** Waste management is continued to be carried out as before, with extensive transports of household waste to incineration in large-scale treatment plants outside the county of Jämtland, and biological treatment of organic wastes only in two communities. There will be only minimal adjustments to ensure that present and future legislation is met. No exemptions from present and forthcoming landfill-bans for combustible and biodegradable wastes are considered.
1. **Digestion.** An anaerobic digestion plant is built in the county centre, to produce biogas for vehicle fuel purposes from organic waste fractions. Other combustible wastes are transported to a large-scale combined heat- and power plant outside the county.
2. **Central composting.** A composting facility is built in the county centre, for treatment of biodegradable wastes. Combustible wastes are transported to a large-scale combined heat- and power plant outside the county.
3. **Local Composting**. Composting facilities are set up locally in each of the eight municipalities of the county, for treatment of bio-degradable waste. Combustible wastes are transported to a large-scale combined heat- and power plant outside the county.
4. **Transition-scenario.** A gradual transition from scenario 0 to 3, via scenario 2 is achieved.

Results from the analysis were presented in a written report (Leander *et al.*, 2003), in an executive summary of this report (Bengard, 2003) and on several oral presentations both for elected local politicians and for municipal officers, in the eight municipalities of Jämtland.

Methods

A questionnaire was set up in accordance with general good practise (Bell, 1987) and sent out by mail to 104 local politicians and municipal officers in the eight municipalities of Jämtland. The respondents were all persons who had taken part in the systems analysis process, or in some way had been informed about its results. The complete questionnaire, included as Appendix A, consisted of five different parts;

1. Background information regarding the respondents, how they participated in the analysis project and how they had been informed about its results (question 1).
2. An individual ranking of the five different scenarios, where the respondent was asked to evaluate each scenario on a five-grade scale from “bad” to “good” (question 2).
3. An individual ranking of different aspects of waste management, where the respondent was asked to indicate on a five-grade scale how important each of these

aspects were to him or her when ranking the scenarios in the previous question (questions 3 and 4).

4. A series of statements regarding the systems analysis and it's usefulness as a decision support instrument, where the respondent was asked to indicate on a five-grade scale to what degree he or she supports each statement. The respondents were also asked to indicate if they had any previous experience of systems analysis (question 5 and 6).
5. Two open questions asking for comments both on the systems analysis and on the questionnaire itself (question 7 and 8).

A reminder questionnaire, identical to the first one, was sent out a few weeks later, to those who hadn't answered. After two more weeks a last reminder was sent and the final number of respondents was after this 66 representing 64% of the original sample. Not all respondents had answered all questions. 7 respondents had given comments on open questions, but only provided few, or no answers, to other questions. The answers were coded into standard software for basic statistical analysis (Microsoft Excel™ with the Analyse-It™-plug-in expansion [Analyse-it Software Ltd, Leeds, UK]). The answers on questions 2, 3 and 5 were coded into discrete numerical values from 1 to 5. Histograms of the distribution of answers to those three questions were analysed, and it was concluded that using statistical methods treating the answers as continuous variables normally distributed around the arithmetic mean value would provide sufficient accuracy. The validity of this conclusion was also tested by analysing the answers to question 5 with both parametric and non-parametric methods. One-sided t-tests were performed to analyse the statistical significance of observed differences in answers to different questions, and χ^2-tests were used to analyse if the composition of group of respondents was significantly different to the composition of the group originally receiving the questionnaire, and in one case to test if two groups of respondents had given significantly different answers on a specific questions. For expression of statistical significance the following probability levels were used; $p \leq 0.05$ (*); $p \leq 0.01$ (**); $p \leq 0.001$ (***).

Results

Background information

In Table 1 the compositions of the original group, and the responding group are compared. The small differences in numbers were not found significant (χ^2-test) indicating that gender and municipal role of the respondents are not significantly disproportional to the original group.

Table 1. Composition of the original population receiving the questionnaire and of the group finally responding. The figures represent the percentage of each category in each group.

	Female	*Male*
Municipal officers		
Received questionnaire	12%	34%
Respondents	16%	36%
Politicians		
Received questionnaire	18%	36%
Respondents	18%	28%
Not specified *(only in respondents)*		2%
Total		
Received questionnaire	33%	67%
Respondents	34%	66%

Other characteristics regarding respondents involvement in the project and how they have learned about the results of the analysis are shown in Table 2. It is noted that female respondents on average have been more active than male respondents, in the systems analysis process.

Table 2. Percentages of the respondents that have been involved in the systems analysis process in different ways (A), and percentages of the respondents that have received information about the systems analysis results in different ways (B).

		Total	*Female*	*Male*
A. Participated in the systems analysis project through	Start-up meeting at Verkön	21%	33%	18%
	Project meeting in Ljusnedal	15%	24%	12%
	Member of steering group	17%	24%	15%
	Have gathered information for the analysis	21%	24%	22%
	Participated in the presentation on September 18	44%	52%	42%
	Other way	17%	19%	18%
B. Have learned about the analysis results by	Reading the full report	45%	43%	52%
	Reading a summary of the report	58%	71%	55%
	Listening to an oral presentation of the results	53%	76%	45%
	Other way	3%	10%	0%

Scenario valuation

The five different scenarios were evaluated on a five grade scale from “good” to “bad” (Table 3). Local composting was found to be the most preferred scenario, and the transition scenario, eventually leading to local composting as well, only slightly less popular. No statistically significant difference was found between scenario 0 and scenario 1, nor between scenario 3 and scenario 4, but all other differences were highly significant (Table 3). Male and female respondents had valued scenarios 0, 2 and 4 differently, but both sexes agreed that scenarios 3 and 4 were the best (χ^2-test; *). Female respondents were less positive to the zero-alternative (average 2.00), than were male respondents (average 2.80) (t-test; **). There was no significant difference on male respondents average ranking of the zero alternative, central digestion and central composting.

Table 3. Average and median score, and final ranking of scenarios, based on the respondents' valuation on average.

	*Average score**	*n – number of respondents*	*Median score*	*Percentile of answers in median and adjacent categories*	*Ranking (based on average score)*
Scenario 0	2.48 [a]	56	2	73%	5
Scenario 1	2.50 [a]	56	2	82%	4
Scenario 2	2.87 [b]	53	3	87%	3
Scenario 3	3.79 [c]	56	4	91%	1
Scenario 4	3.55 [c]	56	4	79%	2

* Values with different indexes (a,b and c) were found statistically different (t-test; $p \leq 0.05$)

Municipal officers and politicians as separate groups came to the same conclusions on which scenarios were the most and least preferable as the complete group of respondents did (t-test).

Important aspects for scenario valuation

In question 3 respondents were asked to indicate on a five-grade scale how important different aspects in the systems analysis were for their valuation in the previous question. The six aspects given the highest and lowest score are given in Table 4. Possibilities for municipal co-operation to minimise costs and negative environmental influence got the highest average score, but statistically equally important aspects are the working conditions for refuse disposal personnel, low emissions of greenhouse gases, households economy and the technical reliability of treatment technologies. Minimising the work efforts for the households and the number of locally and regionally generated job opportunities are considered to be the two least important aspects. It should also be noted that these two aspects are the only ones not reaching an average score above 3.

Table 4. The six aspects achieving the highest (4A) and lowest (4B) average score on importance for valuation of scenarios with significance indexes.

4A. Most important aspects in question 3.	*Average score**	*n – number of respondents*	*Median score*	*Percentile of answers in median and adjacent categories*
P. Possibilities for municipal co-operation to minimise costs and negative environmental influence	4.16 [a]	58	4	100%
G. Sound working conditions for refuse disposal personnel	4.07 [ab]	59	4	98%
C. Low emissions of greenhouse gases such as carbon dioxide and methane	3.97 [ab]	59	4	93%
A. The economy for the households	3.93 [abc]	58	4	91%
F. Known and tested technologies	3.93 [abc]	59	4	93%
E. Minimal transportation work	3.93 [bc]	58	4	95%
4B. Least important aspects				
J. The possibility to recycle nutrients to soils	3.69 [cd]	59	4	93%
L. Low operation costs	3.56 [d]	59	3	86%
I. Low investment costs	3.29 [e]	59	3	92%
N. To what extent the scenarios fulfil existing municipal objectives and plans	3.28 [e]	58	3	90%
K. Minimising the work effort for households	2.95 [f]	59	3	90%
M. Number of locally and regionally generated jobs	2.88 [f]	58	3	83%

* Values with different indexes (a,b,c etc) were found statistically different (t-test; $p \leq 0.05$)

There were differences in how female and male respondents valued different aspects. The six most important aspects according to female and male respondents respectively are given in Table 5. Women have to a larger extent than men emphasised the importance of fulfilling national legislation and objectives, reducing greenhouse gas emissions, and keeping emissions contributing to acidification and eutrophication low. Considering all aspects simultaneously, there is no significant difference between the total average importance factor attributed by female and male respondents (3.79 and 3.63, respectively), so the noted differences are not explained by any group attributing higher values on average than the other, but by actual differences in valuation.

There were also differences between how politicians and municipal officers valued the aspects in question 3. Studied aspect by aspect, there is a difference between how politicians and municipal officers have valued 8 of the 16 aspects (t-tests). The six most important aspects according to politicians and municipal officers respectively, are shown in Table 6. Considering all aspects simultaneously, the total average for municipal officers was 3.53 and for politicians 3.86 (t-test; ***).

Table 5. The six aspects achieving the highest average score on importance for valuation of scenarios by female and male respondents respectively. Also shown are number of respondents (n) and the ranked order within the group based on average scores. T-tests refer to significance test of difference in average valuation between female and male respondents, with differences where p>0.05 considered as not significant (N.S.).

Aspect	*Female*			*Male*			*t-test sign.*
	Average	*n*	*Rank*	*Average*	*n*	*Rank*	
A. The economy for the households	4.10 [ab]	21	5	3.86 [cde]	36	5	N.S.
B. Fulfilling national legislation and objectives	4.24 [ab]	21	3	3.57 [e]	36	12	**
C. Low emissions of greenhouse gases such as carbon dioxide and methane	4.38 [a]	21	1	3.73 [de]	37	6	**
D. Low emissions of acidifying substances and substances contributing to eutrophication	4.24 [ab]	21	3	3.70 [de]	37	8	**
E. Minimal transportation work	3.95 [ab]	21	8	3.92 [cde]	36	4	N.S.
F. Known and tested technologies	3.86 [b]	21	10	4.00 [cd]	37	3	N.S.
G. Sound working conditions for refuse disposal personnel	4.05 [ab]	21	6	4.08 [c]	37	1	N.S.
P. Possibilities for municipal co-operation to minimise costs and negative environmental influence	4.33 [a]	21	2	4.08 [c]	36	1	N.S.

Values with different indexes (a,b,c etc) were found statistically different (t-test; p≤0.05)

Table 6. The six aspects achieving the highest average score on importance for valuation of scenarios by politicians and municipal officers respondents respectively. Also shown are number of respondents (n) and the ranked order within the group based on average scores. T-test refer to significance test of difference in average valuation between politicians and civil servant respondents. Differences where p>0.05 are considered as not significant (N.S.).

Aspect	*Politicians*			*Municipal officers*			*t-test sign.*
	Average	*n*	*Rank*	*Average*	*n*	*Rank*	
A. The economy for the households	4.23 [ab]	26	3	3.74 [cd]	31	7	*
B. Fulfilling national legislation and objectives	3.92 [b]	25	9	3.77 [cd]	31	6	N.S.
C. Low emissions of greenhouse gases such as carbon dioxide and methane	4.26 [ab]	27	2	3.68 [cd]	31	5	**
D. Low emissions of acidifying substances and substances contributing to eutrophication	4.15 [ab]	27	4	3.65 [d]	31	8	**
E. Minimal transportation work	4.10 [ab]	26	5	3.80 [cd]	31	4	N.S.
F. Known and tested technologies	3.9 [ab]	27	10	3.90 [cd]	31	3	N.S.
G. Sound working conditions for refuse disposal personnel	4.00 [ab]	27	7	4.00 [cd]	31	1	N.S.
O. How waste management can be organised in different scenarios, regarding regional co-operation and similar issues.	4.04 [ab]	26	6	3.61 [d]	31	9	*
P. Possibilities for municipal co-operation to minimise costs and negative environmental influence	4.35 [a]	26	1	4.00 [c]	31	1	*

Values with different indexes (a,b,c etc) were found statistically different (t-test; p≤0.05)

This means that politicians on average have valued the aspects in question 3 as more important than the municipal officers, and this difference does partially explain why the significant differences noted between 8 of 16 aspects does not result in a completely different ‘top-list’. The politicians valuations are on a slightly higher level than the municipal officers, but the relative valuations, or ranking order, of different aspects are not quite so different. Both municipal officers and politicians agree that possibilities for municipal co-operation to minimise costs and negative environmental influence is the most important aspect, but politicians attribute higher importance to keeping greenhouse gas emissions, acidifying emissions and emissions contributing to eutrophication low.

When considering ranking order, it also appears that municipal officers value known and tested technologies and sound working conditions for refuse disposal personnel, as relatively more important than politicians do, even though the average scores on these aspects are identical between the groups (Table 6). However, significance indexes (Table 6) show that the differences in average score between the highest ranked aspects within the two groups are only significant in very few cases.

System-analysis as a decision support tool

Under question 5 the respondents were asked to indicate on a five grade scale to which extent they concurred with a number of statements regarding the systems analysis function as a decision support tool. The six statements which the respondents agreed with most are shown in Table 7, and the six statements that the respondents agreed least with are found in Table 8.

Table 7. The six statements about the systems analysis functionality as a decision support tool that the respondents on average agree most strongly with.

Statement (from question 5)	*Average score**	*n – number of respondents*	*Median score*	*Percentile of answers in median and adjacent categories*
L. One conclusion that can be drawn from the analysis is that national objectives and legislation on waste management are poorly adapted to the special conditions in sparsely populated regions.	4.20 [a]	59	4	93%
C. The costs for conducting the analysis are small compared to the costs to carry out the measures that the analysis describes.	3.98 [ab]	58	4	97%
A. The systems analysis is very important as a decision support tool since it allows for an understanding of the full picture and of all consequences of taken decisions.	3.85 [b]	59	4	93%
F. The report has in a good way clarified the conditions for waste management in the future.	3.78 [bc]	59	4	93%
G. The analysis contain enough detailed facts for me to understand what lies behind the different total values.	3.54 [cd]	59	4	92%
P. The results from the analysis support previously available knowledge and thereby strengthen the arguments for decision.	3.51 [d]	59	4	88%

* Values with different indexes (a,b,c etc) were found statistically different (t-test; $p \leq 0.05$)

The difference between the group of statements that the respondents agreed most with, and the group of statements that the respondents agreed the least with was highly statistically significant. That is all responses to the six most popular statements were significantly different from all responses to the six least popular statements (t-test; ***). On this question the gender differences were smaller than in question 3, and both male and female respondents had the same six statements on the list of what they on average agreed with the most. Politicians and municipal officers gave different responses to 8 of the 19 (t-tests; *). Even so, four of six statements are the same for both municipal officers and politicians, both on the list that respondents agree most with, and on the list of statements that respondents agree the least with. Politicians have on average disagreed more strongly than municipal officers on the statement suggesting that the analysis merely confirmed already known facts (t-test; *). Politicians were also more positive to statement O (“the analysis has pointed out what each municipality should do by itself, and on which areas one should co-operate”), where the politicians average score was 3.88 and the municipal officers 2.97 (p=***) Politicians also had a higher average score than municipal officers on statement S (”the analysis has resourcefully highlighted differences in regional and local perspective”), 3.63 and 3.10, respectively (p=*) Municipal officers disagreed more strongly than politicians with statements M (“Long-distance transports had a larger impact on the results than I had anticipated”) and N (“Collection and the emissions resulting from collection had a larger impact on the results than I had anticipated”). This doesn’t necessarily mean that municipal officers had overestimated the importance of these factors to begin with, but it could merely indicate that the results were in level with their expectations.(t-tests; *** in both).

Table 8. The six statements about the systems analysis functionality as a decision support tool that the respondents on average agree with the least.

Statement	*Average score**	*n – number of respondents*	*Median score*	*Percentile of answers in median and adjacent categories*
N. Collection and the emissions resulting from collection had a larger impact on the results than I had anticipated.	2.81 [e]	59	3	93%
M. Long-distance transports had a larger impact on the results than I had anticipated.	2.80 [e]	59	3	81%
B. The systems analysis is so extensive that all sides will find arguments for their already established positions, and therefore it’s value for decision making is limited.	2.80 [e]	59	3	90%
D. The analysis merely resulted in a confirmation of already known facts and did not provide any significant news.	2.78 [e]	59	3	76%
R. Locally generated job opportunities are very important for this type of decisions, and should therefore be addressed in the analysis.	2.64 [e]	59	3	86%
E. Economic aspects are in the end the ones that will decide what measures to take, and the analysis could have been limited to such aspects.	2.32 [f]	59	2	86%

* Values with different indexes (a,b,c etc) were found statistically different (t-test; p≤0.05)

Non-parametric analysis of question 5

To validate that treating the ordinal scale data as continuous variables didn’t impair the quality of the analysis, the answers to question 5 were in parallel analysed with non-parametric statistical methods, designed for ordinal scale data. A Friedman ANOVA test was conducted to rank scenarios based on mean rank (rank sum divided by number of respondents), and Wilcoxon’s W-tests were done to analyse the significance of observed differences (Table 9). The ranking list contains the same 12 statements as were calculated using arithmetic mean values, and the differences between statements were found to be significant in the same cases. The comparison shows that there is no reason to assume that using parametric methods would have impaired the analysis.

Table 9. The six statements about the systems analysis functionality as a decision support tool that the respondents on average agree with the most (9A) and the least (9B), tested with non-parametric methods (Friedman Anova for ranking and Wilcoxon's W-test for significance). The Friedman test requires that only respondents that have answered on all questions are included in the ranking, so *n* was reduced to 56.

9A. The six statements that respondents agree with the most	*Sign. index**	*n – number of respondents*	*Mean rank*
L. …national objectives and legislation on waste management are poorly…	a	56	15.00
C. The costs for conducting the analysis are small compared to the costs…	ab	56	13.76
A. The systems analysis is very important as a decision support tool…	b	56	13.00
F. The report has in a good way clarified the conditions for waste…	bc	56	12.84
G. The analysis contains enough detailed facts for me to understand….	cd	56	11.40
P. The results from the analysis support previously available knowledge…	d	56	11.39
9B. The six statements that respondents agree with the least.			
D. The analysis merely resulted in a confirmation of already known facts…	e	56	7.97
N. Collection and the emissions resulting from collection had a larger…	e	56	7.87
M. Long-distance transports had a larger impact on the results than…	e	56	7.86
B. The systems analysis is so extensive that all sides will find arguments…	e	56	7.60
R. Locally generated job opportunities are very important…	e	56	6.80
E. Economic aspects are in the end the ones that will decide what…	f	56	5.48

* Values with different indexes (a,b,c etc) were found statistically different (W-test; $p \leq 0.05$)

Comments from respondents and other results

Only three respondents had previous experience of systems analysis, so for the vast majority of respondents this study was the first systems analysis they had been involved in. Since the number of respondents with previous systems analysis experience was so low, it has not been considered meaningful to investigate if this experience had any impact on how they answered the questions.

Some respondents utilised the opportunity to give general comments on the system-analysis, or suggestions of aspects that they felt missing in the analysis. Seven respondents express positive comments about the systems analysis (either a satisfaction with the results or a more general satisfaction with how the entire project was conducted). Another opinion frequently expressed (n=5) is that the analysis should have presented the expected results of each scenario for each individual municipality, or at least for the most sparsely populated regions separated from the rest. Other frequent comments (n=5) are related to that it is difficult to answer all questions in the questionnaire due to limited time to process the analysis contents and/or lack of sufficient knowledge about waste management. Four respondents have explicitly expressed need for more information about the results or about waste management issues. Three respondents describe the analysis as too extensive or too difficult to fully penetrate.

Discussion

This paper presents a first approach to evaluate to which extent systems analysis is a useful tool for decision support in waste management. From the responses it is concluded that the systems analysis studied in this case (Leander *et al.*, 2003), was appreciated by both municipal officers and local politicians, and found useful for decision support. From the answers on question 5 about how the respondents perceive the value of the systems analysis for decision support, five of the six statements that the respondents agree most with are statements expressing positive views on the analysis (Table 7), and all of the statements that respondents agree least with are statements that express negative views on the systems analysis, or views related to non-fulfilled expectations on the outcome of the analysis (Table 8). The view that locally generated job opportunities are not very important (Table 4) can find support in other investigations showing that labour intensity in waste treatment is generally low, and seldom result in many jobs (Klang, 2004). There appears to be a high awareness of the importance to ensure sound working environments and conditions for refuse disposal personnel (Table 4), especially among male municipal officers (Tables 5 and 6). This is an important and positive result, since jobs within waste management for a number of years have been considerably more prone to accidents and diseases than most other jobs on the Swedish labour market (Swedish Work Environment Authority *et al.*, 2003).

There are gender-related differences in responses to certain questions. Women have to a larger extent than men emphasised the importance of fulfilling national legislation and objectives, reducing greenhouse gas emissions, and keeping emissions contributing to acidification and eutrophication low (Table 5). Three of Sweden's national environmental objectives are related to greenhouse gases, acidification and eutrophication, respectively, so this co-variation could be expected. However, as Table 2 indicate that female respondents on average have been more involved in the systems analysis process than male respondents, this could also be one reason for the observed differences.

Politicians on average attribute higher values to the aspects mentioned under question 3 than municipal officers do (in 15 of 16 aspects). This could be explained by a general political custom to emphasise the importance of many issues, thereby avoiding irritating or upsetting any interest groups by pointing out certain aspects as less important. The interpretation of this question needs further investigation.

There is a seemingly contradictory result regarding the concern for household economy and their expenses, but apparent lack of concern for investment and operational cost when evaluating treatment scenarios. It should be noted though, that only two aspects get an average importance score below three. Investment costs and operation costs get average scores of 3.29 and 3.56 respectively, so the importance of this 'contradiction' shouldn't be overestimated. Many respondents have also attributed a high importance for municipal co-operation to minimise costs, so the answers could merely indicate that respondents view this as the best way to keep future household costs low.

An apparent contradiction is found in the fact that one of the statements that respondents agree most with, is that the results of the analysis 'support previous knowledge, and thereby strengthen the arguments for decision' (Table 7), at the same time as one of the statements that the respondents agreed the least with, is that the analysis 'merely confirmed already known facts without providing any significant news' (Table 8). This might be explained by a general satisfaction with the system-analysis, and a subsequent tendency to agree more with statements that express positive views and disagree with negative statements. It could also be the case that even though the systems analysis confirmed earlier known facts, it <u>also</u> provided new facts and previously unknown knowledge. Not enough comments have been made by the respondents to draw any conclusion on this matter in the present study.

Respondents of all categories, male and female, municipal officers and politicians, seem to agree that national objectives and legislation in the field of waste management, are poorly adapted to the special conditions of sparsely populated areas. However, it is uncertain if this opinion really is based on interpretations of the studied systems analysis, or on a more general conviction that this is the case. The 45% of the respondents that have read the full report have the same average score on this question as the total group of respondents. Persons who have been part of the so called steering group and have been more involved in the analysis process have an average score on the question that is 4.64, which is higher than the 4.20 scored as average in the total group of respondents (t-test; *).

Some respondents have given comments that the analysis is extensive, and that they would need additional knowledge about waste management, and guidance to interpret the results to be able to answer all the questions in the questionnaire in a meaningful way. Further analysis of how to present the results of a waste management systems analysis should be made.

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Appendix A – The Questionnaire

1. A. **I am a**: ☐ Man ☐ Woman ☐ municipal officer at: ☐ elected official in: ☐ reference-groupmember

(Please note that you may have to cross several boxes)

☐ municipal officer at: → ☐ department responsible for waste management → ☐ other department

☐ elected official in: → ☐ board responsible for waste management → ☐ Municipality board

☐ other: ------------------

B. **I have participated in the project by taking part in:**

☐ Start up meeting at Verkön
☐ Project meeting in Ljusnedal 15-16/4
☐ Steering group
☐ Providing facts for the analysis
☐ The result presentation on the 18/9
☐ Other activity:___________________________

C. **I have taken part of the results from the analysis:**

☐ By reading the written report
☐ By reading the short summary of the written report
☐ By listening to an oral presentation
☐ in another way:_________________________

2. Mark how you personally would rate the scenarios in an actual decision situation.

Scenario	Bad				Good
0. Zero alternative. As today but with the necessary minimum changes to fulfil upcoming legislation.	❑	❑	❑	❑	❑
1. Central large scale anaerobic digestion for the production of biofuel, combined with thermal treatment in external facility.	❑	❑	❑	❑	❑
2. Central large scale composting, combined with thermal treatment in external facility.	❑	❑	❑	❑	❑
3. Decentralised smaller compost facilities, combined with thermal treatment in external facility.	❑	❑	❑	❑	❑
4. The idea description, consisting of a gradual move from scenario 0 to scenario 3 via scenario 2.	❑	❑	❑	❑	❑

Comments:____

3. The systems analysis that you have taken part in addresses many different aspects. Below some of these are mentioned. Please indicate on the scale how important each of these aspect were to you when rating the scenarios above.

Aspect	Not at all important				Very important
A. The economy for the households	❑	❑	❑	❑	❑
B. Fulfilling national legislation and objectives	❑	❑	❑	❑	❑
C. Low emissions of greenhouse gases such as carbon dioxide and methane	❑	❑	❑	❑	❑
D. Low emissions of acidifying and eutrophicating substances	❑	❑	❑	❑	❑
E. Minimal transportation work	❑	❑	❑	❑	❑
F. Known and tested technologies	❑	❑	❑	❑	❑
G. Sound working conditions for refuse disposal personnel	❑	❑	❑	❑	❑
H. High degree of energy recovery	❑	❑	❑	❑	❑
I. Low investment costs	❑	❑	❑	❑	❑
J. The possibility to recycle nutrients to soils	❑	❑	❑	❑	❑
K. Minimising the work effort for households	❑	❑	❑	❑	❑

L.	Low operation costs	❑ ❑ ❑ ❑ ❑
M.	Number of locally and regionally generated jobs	❑ ❑ ❑ ❑ ❑
N.	To what extent the scenarios fulfil existing municipal objectives and plans	❑ ❑ ❑ ❑ ❑
O.	How waste management can be organised in different scenarios, regarding regional co-operation and similar issues.	❑ ❑ ❑ ❑ ❑
P.	Possibilities for municipal co-operation to minimise costs and negative environmental influence	❑ ❑ ❑ ❑ ❑

4. A. Can you think of any aspects missing in the list above that have been mentioned in the analysis. Please state these aspects and how important you regard them to be.

Aspect	Not at all important Very important
	❑ ❑ ❑ ❑ ❑
	❑ ❑ ❑ ❑ ❑
	❑ ❑ ❑ ❑ ❑

B. Can you think of any aspects missing in the analysis, and in that case, how important are them according to you?

Aspect	Not at all important Very important
	❑ ❑ ❑ ❑ ❑
	❑ ❑ ❑ ❑ ❑
	❑ ❑ ❑ ❑ ❑

5. Below follows a number of statements regarding the systems analysis as a decision support tool. Please indicate on the scale on the right to which degree you would agree with each statement.

Statement		Don't agree at all Agree completely
A.	The systems analysis is very important as a decision support tool, since it allows for an understanding of the full picture, and of all consequences of taken decisions.	❑ ❑ ❑ ❑ ❑
B.	The systems analysis is so extensive that all sides will find argument for their already established positions, and therefore it's value for decision making is limited.	❑ ❑ ❑ ❑ ❑
C.	The costs for conducting the analysis are small, compared to the costs to carry out the measures that the analysis describes.	❑ ❑ ❑ ❑ ❑
D.	The analysis merely resulted in a confirmation of already known facts and did not provide any significant news.	❑ ❑ ❑ ❑ ❑
E.	Economic aspects are in the end the ones that will decide what measures to take, and the analysis could have been limited to such aspects.	❑ ❑ ❑ ❑ ❑
F.	The report has in a good way clarified the conditions for waste management in the future.	❑ ❑ ❑ ❑ ❑
G.	The analysis contain enough detailed facts for me to understand what lies behind the different total values.	❑ ❑ ❑ ❑ ❑
H.	Environmental aspects are the most important ones for waste management, and should therefore be the primary focus of a systems analysis.	❑ ❑ ❑ ❑ ❑
I.	The efforts for the households in different scenarios are underestimated in the analysis.	❑ ❑ ❑ ❑ ❑

J.	The analysis provides a satisfactory overview and facilitates comparisons between the alternatives..	❑ ❑ ❑ ❑ ❑
K.	Political and ideological ideas are of great importance when decision makers interpret the results of the systems analysis.	❑ ❑ ❑ ❑ ❑
L.	One conclusion that can be drawn from the analysis is that national objectives and legislation on waste management are poorly adapted to the special conditions in sparsely populated regions.	❑ ❑ ❑ ❑ ❑
M.	Long distance transports had a larger impact on the results than I had anticipated.	❑ ❑ ❑ ❑ ❑
N.	Collection and the emissions resulting from collection had a larger impact on the results than I had anticipated.	❑ ❑ ❑ ❑ ❑
O.	The report clarifies what we should co-operate on, at what is best left to each municipality, respectively.	❑ ❑ ❑ ❑ ❑
P.	The results from the analysis support previously available knowledge, and thereby strengthen the arguments for decision.	❑ ❑ ❑ ❑ ❑
Q.	It is easy to utilise the content of the analysis, even though many different aspects ultimately must be weighted against one another	❑ ❑ ❑ ❑ ❑
R.	Locally generated job opportunities are very important for this type of decisions, and should therefore be addressed in the analysis.	❑ ❑ ❑ ❑ ❑
S.	The analysis have in a good way exposed the differences between a regional- and a local perspective.	❑ ❑ ❑ ❑ ❑

6. This is the first time I have taken part of a systems analysis as the basis for decisions that I will be involved in taking.

❑ *Correct* ❑ *Not correct → Previous analysis concerned:*____

7. Remarks or opinions regarding the systems analysis and how it has been presented.

8. Other remarks (for instance regarding this questionnaire)