Systems Analysis of Small-Scale Systems for Food Supply and Organic Waste Management

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Abstract

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In this thesis, systems for recycling of household organic waste (easily degradable food waste and sewage water) and small-scale systems for food supply were evaluated to see if they could be environment and energy-conserving options. They were evaluated using simulation of static substance-flow models (SFA) combined with life cycle assessment methodology (LCA) for aggregation and interpretation of the results. Three systems were modelled and simulated: i) organic waste management (including transport, spreading on arable land and cropping of grain), ii) bread processing and distribution and finally iii) liquid milk processing and distribution.

The results were found to be very dependent on factors such as choice of system boundaries, transport distances and type of technology. Thus, it was not possible to draw general conclusions regarding the organic waste management system and the scale of food supply system which were most beneficial. However, for the organic waste management system, it was concluded that toilet water-separating sewage systems are a means to increase the rate of nutrient recycling. Furthermore, it was found that urine-separating toilet systems increase nitrogen-recycling rate and decrease energy consumption. The results indicated that anaerobic digestion of organic wastes from society and animal manure could be a system for farmers (or communities) to become more energy self-supporting.

With regard to the food supply system (the transport and processing chain of foodstuffs), it was concluded that energy optimised small-scale food processing and distribution systems could have lower environmental impacts and energy consumption than large-scale systems. However for this to be successful, the advantages of small-scale must be utilised in the entire system, i.e. they should be combined with a nearby local market in order to minimise all transport. The results obtained indicate that the processing step and the private-car transport of food from shop to consumer’s home are the most essential parts of the food supply system with respect to environmental impacts and energy consumption.

Key words: environment, sustainability, energy, systems analysis, modelling, life cycle assessment, LCA, substance flow analysis, SFA, organic waste management, food supply, processing, transport

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Contents

Preface

This thesis was written for the purposes of a Doctoral Degree in Agricultural Engineering. Hopefully you will also find it useful and instructive in your search for more knowledge about technical and natural resources aspects of plant-nutrient (re-)cycling and food supply systems. To non-scientist readers I want to say – Don't think that what you read in this book is the Truth. It is merely an attempt to learn more about the complex things that we call society and reality.

This thesis and the research behind it are my way of trying to contribute to a sustainable development of our society. This is because I am deeply concerned by what our lifestyle and "modern" society bring about in the form of environmental disruptions, depletion of natural resources, too great a dependence on imported resources and weird social structures fostering people who not understand their role in a larger context. My belief is that, if we continue "the race" full-speed straight-ahead as we have done so far, it will end up in a crash, with enormous suffering for the humans in existence at that time. I do not wish that for my children Tor and Björn, or their children, or grandchildren….

Ultuna, August 1999

Olof Thomsson

In theory, there is no difference
between theory and practice, but
in practice there is.

Vint Cerf

De jär skillned pa ti tro u vitä.
Så mikä tror ja i alla fall att ja vait.

Förre julgransförsäljaren A L Skog

It's difference between believing and knowing.
That much, at least, I believe I know.

(The former Christmas-tree salesman Alnus Forrest)

Introduction

This thesis deals with natural resources issues and small-scale food systems. In order to show why and in what context the work has developed, I first present the project organisation and a quite extensive outline of the general background. The general background – including many "soft" humanistic issues – is also important for the understanding of the issues which are of concern, and the potential usefulness of this essay.

An interdisciplinary project

This is the first doctoral thesis produced within the interdisciplinary project "To Integrate Farms and Existing Housing Areas". This project is a joint effort between the Departments of Agricultural Engineering, Ecology and Crop Production Science and Landscape Planning Ultuna at the Swedish University of Agricultural Sciences (SLU). Subjects of major concern in the project are resource management, agroecology, and landscape planning. One licentiate thesis was published in October 1998 (Eksvärd, 1998).

The main hypothesis in this interdisciplinary project was that a closer integration between housing areas and agricultural farms is positive. From an environmental and natural resources perspective, it could for example lead to plant nutrients and organic matter being returned to the soil and environmental impacts being reduced. Furthermore, food transport distances might be drastically shortened and the need for packaging minimised. From a social and pedagogical viewpoint, integration between farms and existing housing areas could play an important role because such integration demands "face-to-face" meetings, both between people and functions in the system. This would, for example, probably lead to better understanding of where food comes from and waste goes to, and to what use the waste could be put – i.e. a better understanding of the real life-supporting systems.

The basic idea of the joint project was to study close integration of geographically-defined housing areas with one or a few specific agricultural farms at up to 3 km distance (walking distance), i.e. local food-supply systems. The general aim was to describe and analyse the effects of, and possibilities for, a closer integration between housing areas and agricultural farms, and to develop tools that could be useful when implementing such integration in reality. The work in the project was carried out through systems analysis using both formalised and descriptive models.

Funding was provided by The Swedish Environmental Protection Agency; The Swedish Council for Planning and Co-ordination of Research; The Faculty of Agriculture, Landscape Planning, and Horticulture at SLU; and The Swedish Council for Building Research.

General background

The driving force for my research arises from a "common-sense" awareness of the fact that the present development of industrialised society is not sustainable. Human society causes ever-increasing environmental disruptions and depletion of natural resources. Furthermore, it is very dependent on non-renewable (fossil) resources. These problems are probably not a threat to life itself – but they pose a significant risk to human welfare and to the existence of our culture.

One important scientific theory, lying behind the fear of catastrophe if we do not change our use of natural resources, is the "pulsing paradigm" presented by H.T. Odum (in e.g. 1994 and 1996). The paradigm in itself does not say anything about catastrophes or environmental problems, but results obtained using this theory may well do so. The paradigm suggests that all developing systems are pulsing – not approaching a steady state as previously suggested by many theories. The theory has been validated against the real performance of many ecological and economical systems. It is also confirmed by the fact that a steady-state type of sustainability may not be possible because short-term advantage favours consumers that use up accumulated reserves. The pulsing paradigm is coupled to the "maximum power principle" discussed later. Figure 1 shows an example of a mathematically simulated pulsing system.

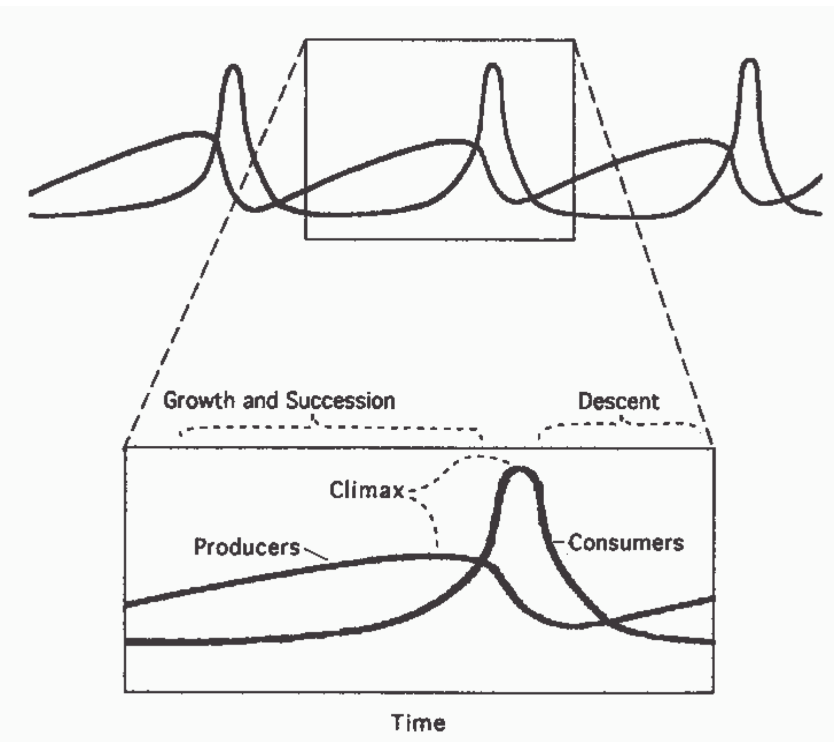

Figure 1. Growth, climax, and descent as part of a longer-range sustainable oscillation. *(after Odum, 1996)*

Long periods with net production of primary products ("capital" or resources) and some growth of consumers are followed by shorter periods of frenzied consumption. After the climax, when the phase of descent starts, both consumption and number of consumers decrease drastically. If we assume that Figure 1 reflects the situation for our society, we are probably in the climax phase. During the 20th Century, much natural capital has been used up to develop growth of economic assets, but in the next stage there may be a rebuilding of environmental resources with a decline in developed assets (Odum, 1996). What will happen to humanity when we approach the phase of descent? Many arguments can surely be raised against this "scenario of disaster", but *the risk* that it can be at least partly true *makes it worthwhile to consider a change in our utilisation of and dependence on resources*. What one can hope is that we find new resources and technologies that flatten out the consumption curve and that the phase of descent not will be as brutal as forecast in Figure 1.

However, there is not just one problem we face, but several. In the following, I will try to give a more structured picture of different aspects of the sustainable development, environmental problems and depletion of resources we need to take into consideration if we want to sustain our society and food production.

Sustainable development; a human-centred concept

Sustainability and sustainable development have been the subject of much debate, maybe because what we define to be sustainable is highly dependent on the context. It should be noted that the subject is development – not growth. Growth is a quantitative increase on a physical scale, while development is qualitative improvement or unfolding of potentialities (Daly, 1990). An example, the author states that "Since the human economy is a subsystem of a finite global ecosystem which does not grow, even though it does develop, it is clear that growth of the economy cannot be sustainable over long periods of time". Furthermore, sustainability is not something which is achieved and then retained. Rather, it is a process with a moving target towards which we ought to strive (Flora, 1994), i.e. sustainable development.

There are many opinions as to whether sustainable development really is possible and, if so, of what it consists. One early definition was given by the Brundtland Report (World Commission on Environment and Development, 1987); "… meet the needs of the present without compromising the ability of future generations to meet their needs". This has been further developed to "a development that secures the material preconditions for the satisfying of today's needs without endangering the material preconditions for future generations to fulfil their needs" (Helmfrid, 1992). These definitions, like many others, are anthropocentric, i.e. it is the living conditions for human beings that have to be sustained and secured. Both these definitions have long time-perspectives. However, it is important to remember that we also have to be sustainable in the short run – we need to survive the day to be able to think about the future. In fact, there are six types of resources that need to be managed properly to maintain sustainability and resilience in any community: natural, individual, social, historical, organisational, and economic (Berg & Nycander, 1997). Thus, short-term economics are important, but should not be the only measure that decides our actions, as is the case to a large extent today.

A broad definition of sustainable developing systems that still comprises many measurable details with the potential for use as indicators for assessment is presented in Eksvärd (1998). The definition consists of three different equally important parts. These concern prerequisites for biological and living systems, prerequisites for human social activities and resource management from a physical point of view ("The Four System Conditions"). The first two parts are presented below and the third part is presented under the heading “Physical sustainability, basis for our existence”.

The systems prerequisites valid for any biological organism or a living system, e.g. societies, have been described by Berg & Nycander (1997). It is argued that every sustainable developing system must be able to: a) support itself, b) adapt to changing situations, c) reproduce both new generations as well as knowledge and culture, d) maintain its boundary functions; i.e. both preserve identity and allow interactions, and e) control the system; e.g. make laws and plans, maintain a market and stimulate social self-organisation.

Since most of what we think of concerning sustainable development is anthropocentric, the sustainability of the resource base makes little sense if it is separated from the human agents who manage the environment (Redclift, 1993). Thus, it should be appropriate to include the prerequisites for human social activities, i.e. the human needs, in the definition. Max-Neef (1989) has identified nine basic unexchangeable human needs: a) material support, b) security, c) appreciation, d) participation, e) creativeness, f) identity, g) freedom, h) relaxation, and i) meaning. All these should be fulfilled if a society claims to be sustainable. Of course these "demands" have to be interpreted with common sense, and differently in different situations and cultures, but in general they are relevant.

Results from a pilot study presented in Eksvärd (1998) may give an indication of how important it is to have ordinary people engaged in the process of sustainable development. An assessment was conducted of the factors affecting and steering all physical flows in a rural settlement. It was concluded that personal choices of citizens decide, or at least affect all of them (Paper 1 in Eksvärd, 1998). Weather might be an exception but the burning of fossil fuels affects even that. Choices of technology and consumption habits turned out to be the most frequently-occurring factors which decide the physical flows, i.e. the physical sustainability of the society. This shows the importance of focusing on human acceptance and participation in a process of change that affects each and everyone’s daily life. Furthermore, to achieve sustainability, changes in attitudes and behaviour will be required at all levels of society (Jacobson et al., 1996). The importance of getting people involved in the sustainable development process is also stressed in the Agenda 21 document agreed on at the UN conference on environment and development in Rio de Janeiro, 1992 (Agenda 21, 1992).

The involvement of citizens in the development process inevitably also means that they have to co-operate in order to organise the development strategy. One important characteristic for well-functioning organisations on all levels is embedded in the word *trust* (Gibb, 1978). The so-called Gibb triangles (Figure 2) have developed out of the ideas in this book entitled *Trust – A New View of Personal and Organizational Development*.

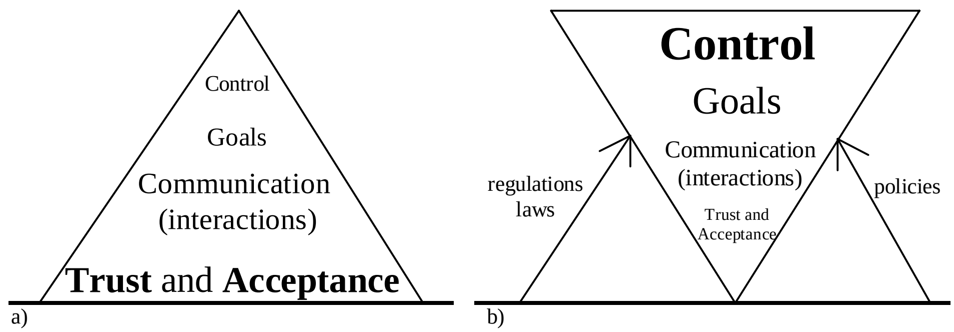

Figure 2. Different strategies of organization: a) Emphasis on Trust and Acceptance; a more self-stabilising system and b) Emphasis on Control; a system that needs laws, regulations and policies to be stable. *(idea from Gibb, 1978)*

The triangles are an attempt to illustrate the balance between activities needed to make organisations function well. Organisations are human activity systems, which are fundamental in any discussion about sustainable development. The a) triangle describes the ideal situation: An organisation that relies on trust and acceptance and has well-developed communication and interactions. Goals are needed and also a degree of control, but not too much. This is a stable system standing steady on the ground. The b) triangle shows how organisations and our societies often are organised. The emphasis is put on control and common goals. Some efforts are made on communications and interactions but too little. Trust and acceptance have a very small place. This system is more unstable and needs extensive quantities of laws, regulations and policies to be kept running. However, in reality, there is always a need for policies, laws and regulations and thus the optimal system is probably somewhere between the two triangles.

Physical sustainability, basis for our existence

To create a sustainable society, we need to co-operate and have everyone taking part in the process but *ultimately physical conditions are the basis for our life*. It is the earth with its resources and ecosystems we need to nurture and take good care of. Thus, knowledge about physical systems is vital to be able to accomplish sustainable development. I will try to explain some different aspects of physical sustainability below.

Life-supporting systems

The ecosystems covering the world play an important role in primary production, air and water cleaning, noise reduction, waste assimilation and so on. It is these life-support systems that make it possible for humans to breathe, drink and eat. The problem is that we tend to take all this work for granted because we don't pay any money for most of it (Odum E P, 1987). Thus, knowledge about how our activities affect these systems is essential to prevent humanity degrading the ecosystems vital for our wellbeing.

Nowadays, it is *non-point pollution* (originating from scattered or diffuse sources such as car exhausts and runoff from agricultural fields) that places the greatest stresses on the global life-supporting environment. We have managed to achieve a significant reduction in point-source pollution (entering the environment through e.g. chimneys, pipes, and ditches), though much still remains to be done. This is probably because such pollution can be managed at the output side (at the end of the pipe). The non-point pollution is harder to manage, since it can only be controlled by managing the inputs to production. E P Odum (1987) calls this Input Management.

This has been further developed in four basic principles for physical sustainability called "The Four System Conditions" by Holmberg (1995) in collaboration with the Swedish Foundation "The Natural Step" and a large group of scientists, organisations, companies etc. (Natural Step, 1999). Those principles are the following:

1) Substances from the earth's crust must not systematically increase in nature.

In a sustainable society, human activities such as the burning of fossil fuels and the mining of metals and minerals would not occur at a rate that causes them to systematically increase in the ecosphere. In practical terms, the first condition requires society to implement comprehensive metal and mineral recycling programmes and to decrease economic dependence on fossil fuels.

2) Substances produced by society must not systematically increase in nature.

In a sustainable society, humans would avoid generating systematic increases in persistent substances such as DDT, PCBs, and freon. Society needs to find ways to reduce economic dependence on persistent human-made substances.

3) The physical basis for the productivity and diversity of nature must not systematically be diminished.

In a sustainable society, humans would avoid taking more from the biosphere than can be replenished by natural systems. In addition, people would avoid systematically encroaching upon nature by destroying the habitat of other species. Society's health and prosperity depends on the enduring capacity of nature to renew itself and rebuild waste into resources.

4) We must be fair and efficient in meeting basic human needs.

Meeting the fourth system condition is a way to avoid violating the first three system conditions for sustainability. Considering human enterprise as a whole, we need to be efficient with regard to resource use and waste generation in order to be sustainable. Achieving greater fairness is essential for social stability and the co-operation needed for making large-scale changes within the framework laid out by the first three conditions.

As can be concluded from the four system conditions, it is not the emissions or consumption of resources *per se* that are a threat to the life-support system, it is their magnitude. During recent years, it has been proposed that the Western industrialised countries must decrease their consumption of resources by a "Factor 10", i.e. by 90 percent. This does not mean that the use of all materials should be decreased to such a large extent, but the most harmful (to humans and environment) might have to be completely abandoned, while others can be used at a less drastically decreased level (Gardner & Sampat, 1999).

It is often claimed that the production of items such as e.g. cars, packaging and telephones are more efficient concerning the use of resources per unit produced than ever before, which is thought to lower the total consumption of resources. However – consumption is still growing due to a "growing demand" for ever more products (Gardner & Sampat, 1999). This growing demand can partly be explained by the worldwide emergence of a "consumer-class", i.e. people with high enough income to afford a consumption pattern which demands a high level of throughput of materials and energy. They own cars, TVs, dishwashers, mobile phones, etcetera, etcetera. This class is also identified by the fact that consumption itself has become a social measure (status) of success (Carlsson-Kanyama, 1997). This might also explain why all efficiency improvements in e.g. car engines so far have mainly been utilised in the provision of more power to achieve higher speeds, not decreased consumption of resources. Changing this pattern of living is probably vital, but it is regrettably not within the scope of this work.

Depletion of fossil energy resources

The development of the industrialised society is largely founded on the discovery of fossil energy sources such as coal, oil and gas. Today we are more dependent than ever on these energy carriers. In Sweden, about two thirds of the total energy supply comes from fossil energy sources (oil, gas, coal, uranium) (calculation from SwNEA, 1998). However, it should be noted that Sweden produces almost half of its electricity by hydropower (counted as renewable). Many other countries are even more heavily dependent on fossil energy. For transportation, which is the fastest growing energy use sector globally (Mårtensson, 1997), fossil fuel is totally dominant.

With regard to these fossil energy resources, one can conclude that even if the existing amounts of oil, natural gas and coal were excessive they should not be utilised, due to the high risk of climate change and other ecosystem disturbances discussed above. Furthermore, and in the short-terms equally important, is that even if the sources of fossil energy are vast, these fuels will probably become much more expensive within a decade or two (Agfors, 1996; Hatfield, 1997a; Nilsson, 1997). One important reason is an increasing dominance on the market of the OPEC-countries. Another reason is that global oil production capacity is at, or is very close to, its historical maximum. Discovery and exploration of "new" oilfields are not keeping pace with production. However, the Energy Information Administration of the US Department of Energy (DOE/EIA) writes: "Oil prices are expected to remain relatively low, and resources are not expected to constrain substantial increases in oil demand through 2020." But this prediction assumes that: "On the other hand, the reference case projection anticipates that as much as three-fourths of the increase in demand over the next two decades will be met by increases in production by members of OPEC rather than by non-OPEC suppliers. …. Some analysts suggest that OPEC might prefer and pursue significant price escalation through conservative capacity expansion decisions rather than undertaking such an ambitious production expansion effort. The view presented in this outlook discounts such an expectation." (DOE/EIA, 1998). Furthermore, nothing is mentioned about the 40- to 50-year perspective in the DOE/EIA outlook. Other fossil resources such as tar sands, heavy oils and oil shale could possibly be refined to vehicle fuel. The problem is that the development of refinery processes will probably not be online by the time (2010-2015) they are needed (Hatfield, 1997b).

An important factor concerning the use of natural resources, especially fossil energy reserves, is the environmental movement and public opinion. This will surely try to force politicians to put high surcharges on the use of coal in particular but also on fossil oil and gas and other non-renewable energy resources. In Sweden at least, awareness of environmental issues among a large part of the population and politicians has grown quite strong during recent years. Much of this is due to broad programmes and discussions spurred on by the Agenda 21 document agreed on in Rio de Janeiro 1992 (Agenda 21, 1992).

Altogether, this might lead to society not having access to cheap fossil energy – or not being allowed to use it. Thus, *it is of the utmost importance to plan for a genuine shortage of fossil oil now*, rather than waiting for it to happen before we react, as we did in the 1960s and 1970s. To be able to manage the situation, we probably have to both lower energy consumption and to develop technology for the utilisation of renewable and environmentally less disturbing energy sources or energy carriers, e.g. solar cells, hydrogen and fuel cells and biofuels such as biogas and ethanol.

Depletion of natural resources through linear material flows

Other non-renewable natural resources such as metal and mineral ores are also consumed at a much higher rate than they are re-created. Most scientists seem to agree that the use of these fossil resources will eventually have to be very limited but opinions on how fast the change must be accomplished vary greatly. However, in contrast to fossil energy sources, natural resources used as materials can be reused and recycled. It is also often possible to substitute other metals and materials for them.

Not surprisingly, it was concluded in a pilot study comparing two sites of different size and location (one large on the outskirts of a city and one small rural village), that most physical flows go through the systems, starting in the far surroundings and ending somewhere else in the far surroundings (Paper 1 in Eksvärd, 1998). This might be called linear flow management, in contrast to cyclic flow management where the flows are reconnected. No substantial differences in this aspect were found, although the small settlement had some re-connected flows, i.e. the people living in the countryside are also part of the linear-flow-society. However, one has to remember that when studying material flows it is very important to carefully consider the choice of system boundaries to obtain a proper scale of the system studied, because in one scale or time-horizon all flows in the system appear linear and in another scale or time-horizon many of them appear cyclic. Natural processes such as geohydrological processes and plant-respiration often disperse matter over a wide area; i.e. the feedback process takes place in a large part of the biosphere. These flows appear linear when studied on a local scale but are in fact reconnected. However, the time-span for these processes is often long.

It should also be remembered that our immense resource consumption is not at all strange, because when resources are abundant it is competitive to use as much as possible. However, when resources become scarce and limiting for the performance of the system, it is efficient to re-cycle and re-use them (Jordan, 1996). This linear resource management is probably a heritage from the time when man believed in unlimited sources of e.g. oil and metals. Now when it is recognised that that is not the case anymore, we have to reorganise our systems to be cyclic in order to use sparse resources efficiently.

The need of feedback as reinforcement in systems is supported by the "maximum power principle" presented by H T Odum (in e.g. Odum, 1987 and 1996). This author has developed a concept presented by Lotka in the 1920s (Lotka, 1922a; 1922b). It suggests that all self-organising systems tend to maximise the use of power, because systems which process more useful energy will succeed other systems since they have more energy to handle unforeseen events and better adaptation to changing surrounding conditions. Systems that are designed with "autocatalytic" feedback (the production in the system is the product of input sources and feedback from storages produced in the system) maximise power. This can be explained by the fact that all energy transformations have less energy in the output (according to the Second Law of Thermodynamics), and to prevail, this lesser energy produced or stored must be able to feed back and reinforce the input production. This is only possible if the feedback multiplies or otherwise amplifies the process. Thus, energy transformations that do not develop the reinforcing design will not be reinforced nor long continued (Odum, 1996), i.e. they will not be sustainable.

Resources and food-production

Resources essential for agricultural production are water, soil suitable for cropping, plant nutrients (as e.g. nitrogen, phosphorus, and potassium), and energy (for cultivation, weed control, and harvesting). In industrial conventional agriculture, the energy and nutrients used are often of fossil origin.

Estimations of the remaining available amounts of these differ greatly and thus the estimated time left for consumption varies a lot. For oil, predictions say at least between 40 and 200 years (Hatfield, 1997a; DOE/EIA, 1998) and for phosphorus between 200 years (reserves) and 1200 years (estimated geological reserves) (Louis, 1993). The resources of potassium are rarely mentioned (probably since it is not recognised as a water pollutant) but potash reserves mineable with current technology and current infrastructure are estimated to last approximately 250 years at current consumption levels (Louis, 1993). Potential reserves (identified resources which demand investments in infrastructure and possibly technological research) would suffice for 1000 years of present consumption level. Anyhow, this dependence on imported resources (long lasting or not) causes a structural vulnerability in agriculture. It is argued that existing Swedish and European agriculture is very vulnerable due to e.g. specialisation, long transport distances and heavy dependence on cheap, ever-flowing fossil fuels and nutrients for crops and animals (Günther, 1995). Thus, from both an environmental and economic point of view there is *clear motivation to start planning for a future society that is less dependent and very efficient in the use of fossil resources*.

Use of organic wastes as fertilisers in agriculture might be a method to decrease the dependence on imported resources and to use existing resources more efficiently. It could also be a means to strengthen the feedback functions in the system. For quite a long time, recycling of easily degradable organic waste has been discussed and carried out to some degree but, up to now, the recycling of phosphorus in particular has been a matter for concern. This might seem a bit odd, since cheap energy, and following that, nitrogen, are likely to be the first resources to become expensive. The explanation is that phosphorus is recognised as a major water pollutant at the same time as it is quite easily trapped by chemical precipitation. Nitrogen is also a water pollutant but is harder to catch and recycle.

Another strong argument for organic waste recycling is soil organic matter. In many regions of intense cropping (without animal production and grassland farming), which rely on chemical fertilisers, there are problems with decreased humus content in soils. Organic fertilisers could be a means to prevent increased soil erosion. In regions with fertile clay soils, like the plains in southern Sweden, soil erosion is not a matter for concern, but decreased content of organic matter in the soil results in deterioration in soil structure and physical properties. Clay soils with mainly annual crops are estimated to be able to produce 10-20 % larger yields of grain if their structure and physical properties are restored to the level they had 40-50 years ago – or today's yields could be obtained with use of less inputs. Better soil structure would also decrease the need of tractive power, and thus energy consumption, by maybe 15-20 % (Johansson et al., 1993). However, organic fertilisers could also carry a risk for increased environmental hazards. Larger content of organic matter in soils may lead to amplified eutrophication if proper cropping practices are not used.

In addition, human organic wastes could be an important plant nutrient source for organic farmers, to compensate for nutrients exported in products sold. This is not possible today due to regulations in, for example, the EU, but if methods for such resource utilisation are proven to be proper and hygienically safe, the rules will surely be changed. Besides the agricultural need, this also could be a means to solve some of the waste disposal problems in society, or rather to change it from a waste problem to a question of resource utility. These arguments should, logically, have the effect of making recycling (reuse) of plant nutrients and organic matter a more important issue on the agenda again.

Systems thinking and systems analysis

As discussed in “General background”, it is clear that both natural science knowledge about resources consumption and material flows, as well as knowledge in human sciences, together with people's comprehension and enthusiasm are needed to accomplish sustainable developing food supply systems. For, as Allen et al. (1993) so clearly expressed it: "A sustainable food and agriculture system is one which is environmentally sound, economically viable, socially responsible, nonexploitative, and which serves as the foundation for future generations".

What seems to be lacking is knowledge about how to accomplish a turnaround in the development and, maybe even more, the conviction that our changes will have the desired effect. The latter is, I think, a healthy attitude to ensure that the measures carried out become improvements and not just changes. Following the "maximum power principle" (discussed above) it is obvious that organisms, species, cultures, etc. that want to keep a place in the system (be sustainable) have to feed back reinforcing energy (in a broad sense) to the system. That implies recycling of e.g. organic material but – and it is an important but – it also implies that the recycling has to promote the entire system. Recycling in itself is of no obvious value and, if it does not have a reinforcing function for the system, it will not be competitive in the long run either.

In other words, it is knowledge about the entire system that is needed. Such knowledge can only be obtained by use of different systems analysis methods in combination with co-operation between different sciences and competencies. However, there are at least two different types of systems research method. Checkland (1989) describes the two as *hard* and *soft* systems methods. The term systems analysis often refers to hard systems analysis, developed by engineers to answer questions that have clear and rational causes and/or clear answers. The questions are of the type, *how* do we act to make a process work efficiently or how do we construct a rocket that can manage to land on the moon? Soft systems methodology handles problem situations where human activities are involved and where neither questions nor answers are clear – situations of uncertainties. The questions are more of the type, *what* is making a group of people not work well together? The soft systems approach identifies both problems or questions that are best handled by hard systems analysis methods and problems that should be dealt with in a soft systems approach. Starting with a hard systems method often has the result that the soft questions are forgotten. This is one of the reasons for inclusion of this quite extensive chapter about the general background.

For a research method or development strategy to be able to make sustainable development operative, it has to manage evolutionary perspective at same time as it gives space for systems thinking (Helmfrid, 1992). This might be called holism (Figure 3). Conventional research, which influences the thinking in the entire society, is often found in the "other corner" of the diagram of thinking and perspectives.

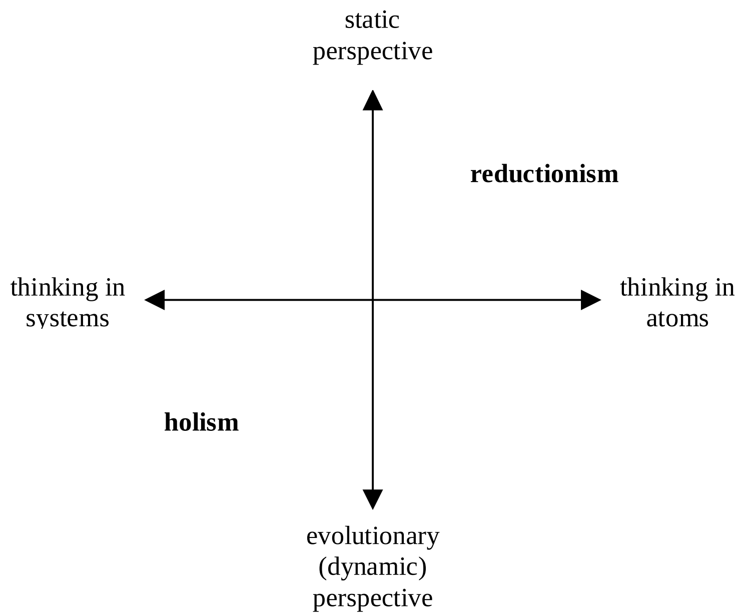

Figure 3. Dimensions of thinking and perspectives. *(after Helmfrid, 1992)*

The term evolutionary refers to a perspective permeated by the consciousness that every change in the system will create other changes – everything is part of a larger context that evolves. The work presented in this thesis is definitely found to the left in the diagram but probably more against the upper (static) sector, although an evolutionary perspective is included.

Systems analysis methods

Since "the environmental problem" has many dimensions, multi-dimensional tools have to be used. There are several methods under development to assess the environmental performance of products, processes, systems, organisations, societies and so on. The different methodologies have somewhat different purposes, which I will not comment on any further here. What most of them have in common is the cradle-to-grave perspective, i.e. all impacts from the extraction of raw materials to the management of waste are included. The perhaps most developed method is Life Cycle Assessment (LCA). Examples of others are Ecological Footprints Analysis, Environmental Space Concept, Environmental Impact Assessment (EIA), Substance Flow Analysis (SFA), Material Flow Analysis (MFA), Environmental Accounting, Foodshed Analysis, and EMERGY analysis.

It is, however, not only the question of multi-dimensional problems we have to deal with. Since a systems perspective is needed, it also implies that we have to handle very complex systems. To be able to analyse such (complex) systems, computerised mathematical modelling is a powerful tool. Much knowledge is gained simply by the fact that, to be able to construct sufficiently detailed models, one has to collect good data about the system's structure and performance. This kind of model also makes it possible to test e.g. different combinations of sub-systems, without expensive construction of the real systems.

The mathematical computer models of different systems used in the work presented in this thesis are a type of hard system analysis combining SFA and LCA methodologies. Thus, they handle both the complexity and the multi-dimensionality. The models are used for simulation of systems with more or less well defined properties and behaviour.

Scope of the work

This thesis deals with natural science (hard systems) aspects of cyclic small-scale food systems. A graphical representation of a cyclic food system as I see it in this work is presented in Figure 4. Dotted lines indicate the system boundaries for the study.

The work is presented as three separate studies of different parts of the food system. First comes a study of organic waste management. This is an important part in the system, concerning both environmental aspects and how to get the system more self-dependent for plant nutrients and possibly energy. Included in this part are also: grain production resulting from the recycling of organic wastes, nitrogen emissions from soil, energy consumption and emissions from grain-production field operations, and manure from fifteen dairy cows including young stock, i.e. the parts of the agricultural system with direct connections to the organic waste management system.

The second (bread production system) and third (liquid milk production system) studies concern food processing and distribution, which are also vital parts, especially concerning energy demand and environmental impacts emanating from the burning of fuels and production of electricity.

Other parts of concern that have not been included are the major parts of the primary production (agriculture) and the consumption of the food (consumer phase). These are of importance for environmental impacts and consumption of resources when the whole food system is discussed, but do not change between scenarios of the systems studied here.

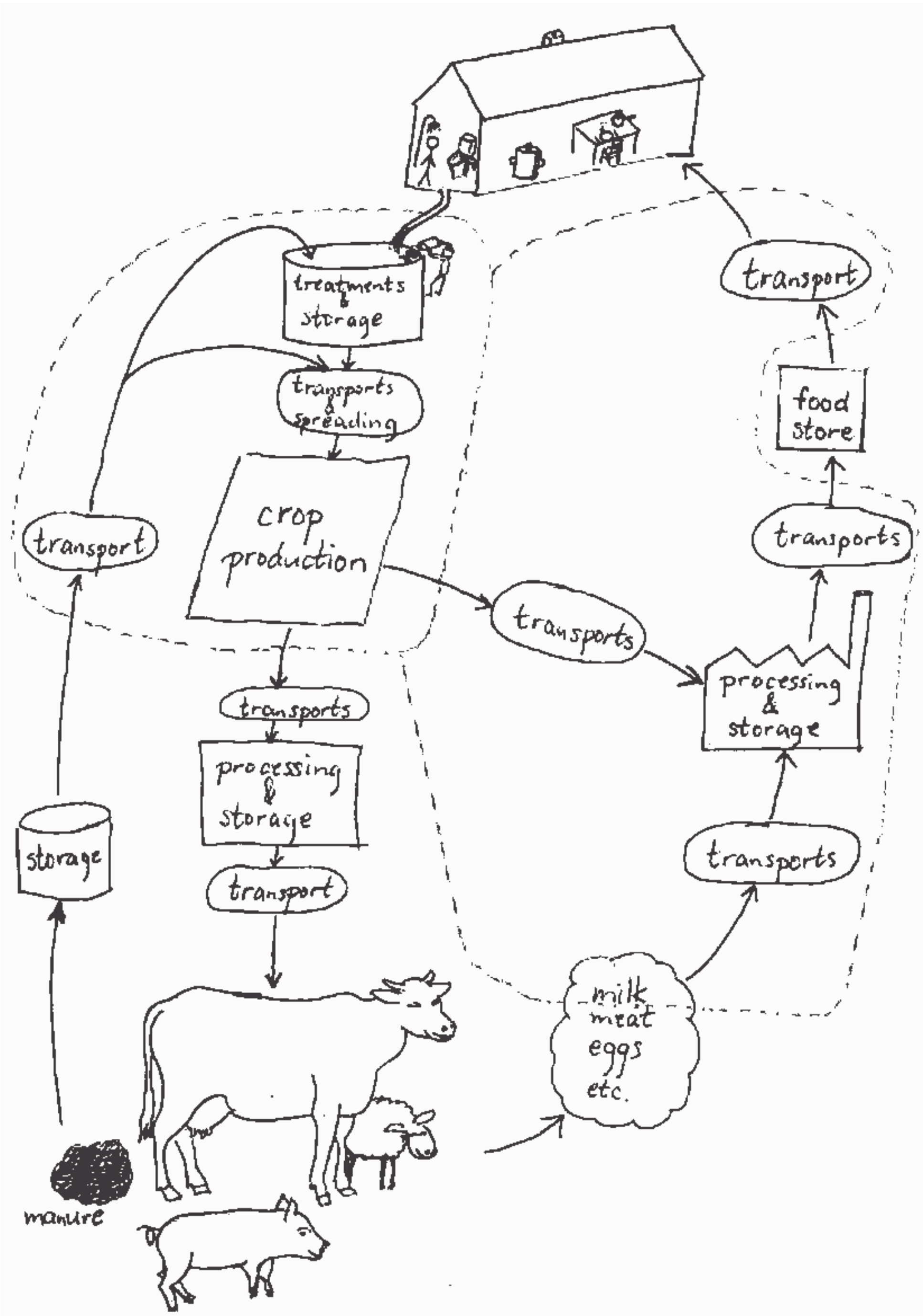

Figure 4. Schematic representation of the parts in a cyclic food system.

Aims and objectives

The general aim of my work has been to learn more about natural science (physical sustainability) aspects of cyclic and small-scale (locally based) food systems and especially the systems that connect agriculture and consumers in both directions (Figure 5).

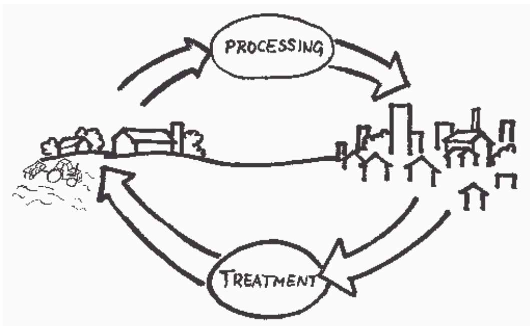

Figure 5. Schematic picture of a cyclic and local food supply system.

The general objective was to:

- investigate if development towards cyclic and small-scale food systems could be a means to decrease environmental impacts and resource consumption
- develop systems analysis tools that can be used in further case studies and in planning of real-world sustainable development projects.

Specific aims and objectives of the organic waste management and grain production systems

The primary issues of concern in this part of the work are recycling of plant nutrients to arable land, expected additional yield of grain resulting from the recycling, energy turnover, and environmental impacts.

The objectives of the organic waste management part of the work were:

- to establish knowledge about the flows of plant nutrients and heavy metals in small-scale handling system for household wastewater and organic wastes
- to evaluate the importance of flows of household residues compared to the flows of organic residues in agriculture
- to evaluate to what extent these flows can be of importance for agriculture
- to compare different organic waste treatment systems concerning environmental impact and consumption of natural resources
- to find efficient (in terms of resource utilisation) recycling systems, which thus contribute to producing much food and at the same time are energy self-supporting or, even better, net energy-producing

Specific aim and objectives of the bread processing and distribution system

The aim of the bread production part was to compare large-scale and small-scale food processing systems and the transport connected to them from an energy and environmental point of view.

The objective of the study of bread production systems was:

- to identify important parts in the systems, and to assess their importance for the whole system
- to establish if long-distance transport is as important as many want to believe
- to evaluate if internal energy efficiency in large-scale facilities is as important as others claim

Bread was chosen because it is a staple food that is produced all year round and can easily be produced on different scales. Furthermore, local as well as imported raw products can be used.

Specific aim and objectives of the milk processing and distribution system

An introductory study of milk processing and distribution was performed in order to give knowledge comparable to that obtained in the bread production study. The objective was to create a broader picture of the system for food processing and to assess the generality of the results obtained in the bread production study. Thus, the same system boundaries as in the bread production study were used in this study.

Milk was chosen to represent a different type of staple food. It is not easily stored, it contains a lot of water (i.e. more energy demanding to transport) and the large-scale advantages in its processing are presumed to be larger.

Methodology

Cyclic food-supply systems are large and complex. They have many interactions both between parts within the system and with other systems. Thus, a systems analysis approach is necessary to analyse such a system. Figure 6 shows a conceptual representation of the properties of the system under study.

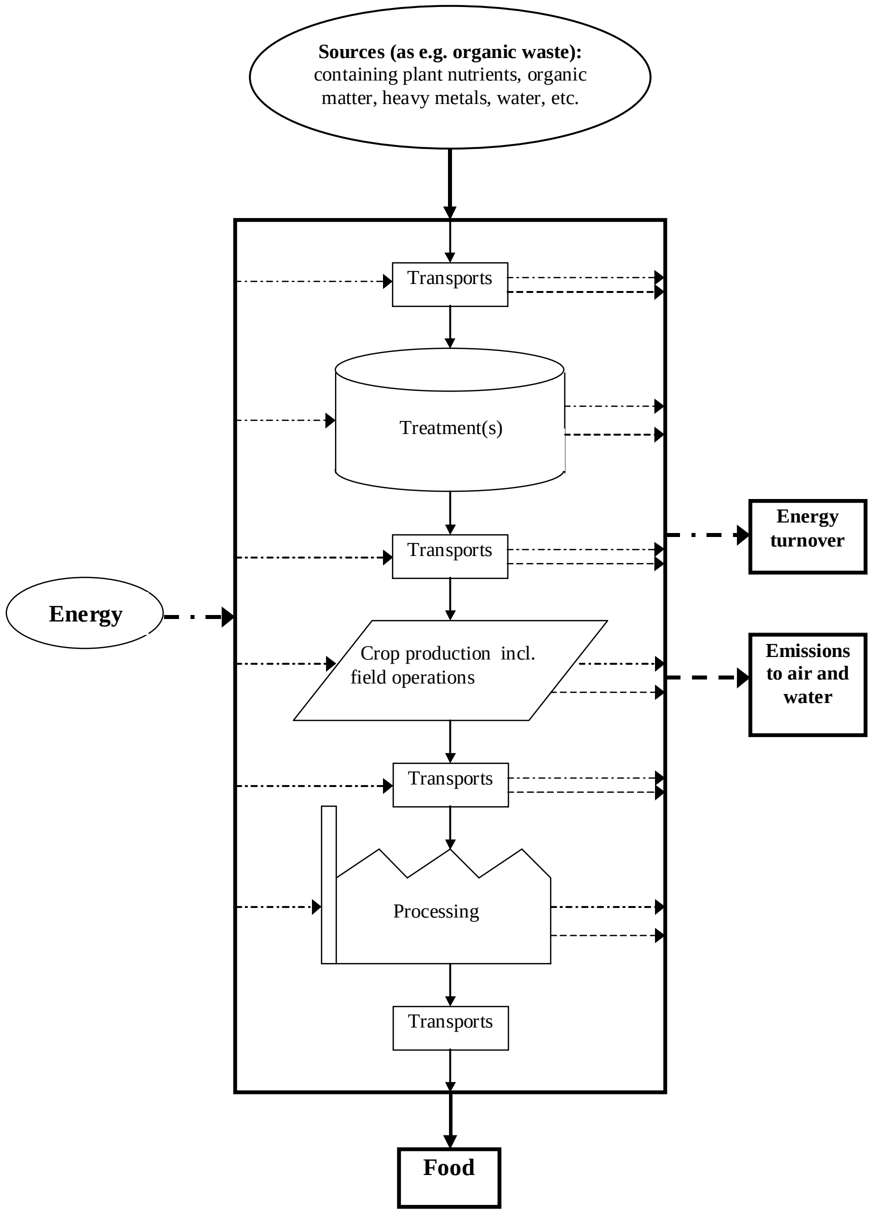

Figure 6. Conceptual representation of a cyclic food system from a energy and environmental emissions point of view.

When the issues of concern are quantifiable, mathematical computerised models are powerful tools. The model approach makes it possible to perform experiments on whole systems without (or before) constructing them in reality. It also gives an opportunity to compare the performance and importance of different parts in the context of the entire system. Furthermore, it is possible to make sensitivity analyses and thus investigate if and how changes in parts of the system will affect the total. Since one of the objectives with the study was to determine the flows of plant nutrients and organic matter etc., a substance flow analysis (SFA) model was needed. For the environmental impact assessment, other methods such as e.g. life cycle assessment (LCA) were needed. A SFA model for assessment of municipal organic waste and wastewater management, which may be combined with several elements of an life cycle assessment, has been developed in Sweden (Dalemo et al., 1997). The concept of this model, called ORWARE (ORganic WAste REsearch) was found to suit the purpose of the present study well.

Thus, the work presented in this thesis has been carried out through mathematical modelling and simulation following the ORWARE modelling concept. The models are modular, i.e. each part of the real system is described in sub-models that may be combined to models of different systems. The researcher chooses sources, treatments or processes, transport and spreading or distribution technology. The models are linear and mainly static. No optimisation is carried out in the models. Each sub-model calculates the degradation of its incoming material, formation of new substances, division of the material flow(s) into several outputs (e.g. sludge, gases and outlet water), and energy turnover. Model outputs are yearly averages of sub-processes and the entire system respectively. The models are programmed in the software Matlab/Simulink (The MATHWORKS Inc., 1997).

The models used in this study are thoroughly described in the chapters "Organic waste management system", "Bread production system", and "Milk production system".

Like the original ORWARE sub-models, those developed here had a vector of 43 positions, i.e. 43 different substances (or in some cases, groups of substances) were included in the calculations. However, due to lack of data in several processes, only some of these positions were used in this study. For example, it was not possible to include polyaromatic hydrocarbons (PAH) and adsorbable organic halogens (AOX). The positions used in the outputs are described in Table 1.

Table 1. The output variables.

Abbreviation	Remarks
N-tot	Total nitrogen
$N\text{-}NH_3/NH_4^+$	Nitrogen in ammonia and/or ammonium
$N\text{-}NO_3^-$	Nitrogen in nitrate
P	Phosphorus
K	Potassium
CO_2-f	Carbon dioxide of fossil origin
CO	Carbon monoxide
CH_4	Methane
VOC	Volatile Organic Compounds, e.g. un-combusted hydrocarbons from engines
$N\text{-}NO_x$	Nitrogen in nitrogen oxides
$N\text{-}N_2O$	Nitrogen in dinitrogen oxide (laughing gas)
$S\text{-}SO_x$	Sulphur in sulphur oxides
Cl	Chlorine
Pb	Lead
Cd	Cadmium
Hg	Mercury
Cu	Copper
Cr	Chromium
Ni	Nickel
Zn	Zinc
C-total	Total carbon
COD	Chemical Oxygen Demand
BOD_7	Biological Oxygen Demand during 7 days
VS	Volatile Solids (=DM minus ash)
DM	Dry Matter
H_2O	Water

Assessment of resources consumption and environmental impact

The environmental impact was assessed using LCA methodology. First, all inputs and outputs were classified, i.e. grouped in environmental impact categories. Each substance may fall into one or several impact categories. Step 2 was the characterisation, where the relative contribution of each substance within the impact category was assessed and all contributions in the category were aggregated. Lindfors et al. (1995) propose 15 impact categories (Table 2) that should be reported, or at least commented on. These can be further divided into sub-categories.

Table 2. List of impact categories *(Lindfors et al, 1995)*

Impact categories	
1.	Resources – Energy and materials
2.	Resources – Water
3.	Resources – Land (including wetlands)
4.	Human health – Toxicological impacts (excluding work environment)
5.	Human health – Non-toxicological impacts (excluding work environment)
6.	Human health impacts in work environment
7.	Global warming
8.	Depletion of stratospheric ozone
9.	Acidification
10.	Eutrophication
11.	Photo-oxidant formation
12.	Ecotoxicological impacts
13.	Habitat alterations and impacts on biological diversity
14.[#]	Inflows which are not traced back to the system boundary between the technical system and nature
15.[#]	Outflows which are not followed to the system boundary between the technical system and nature

[#] Not impact categories but should be included

Impact categories included

I chose to include six of the impact categories (or parts of them). These were: consumption of primary energy and water resources, heavy metal contamination of arable land (ecotoxicological impact), eutrophication, global warming, acidification and photo-oxidant formation. The first three were chosen since they may differ between the different scenarios studied. The latter three are largely an effect of energy consumption, chosen since that is the major difference between scenarios in primarily the food production (bread and milk) study.

Primary energy resources

To be able to make a comparison between scenarios which use much electricity with those mainly using fuels, energy consumption was traced back to the consumption of primary energy resources. This measure was an attempt to consider the resources consumption in the lifecycle of the energy carriers. It did not evaluate the differences in energy quality. There are other measures, e.g. exergy, embodied energy, and EMERGY, which do, but it was not possible to include these in this study.

The primary energy consumption in production of average Swedish electricity demand is shown in Table 3. Transmission losses in the distribution net (7 %), pre-combustion energy consumption for fuels and efficiency in e.g. hydropower and nuclear power are included, making 2.35 MJ primary energy per MJ electricity used. The equivalent value for electricity produced in oil-fired power plants is 2.69 (Habersatter et al., 1998). For reasons of simplicity, I used an average for different fuels; 1.2 MJ primary energy per MJ fuel (calc. from Tables 16.4 and 16.9 in Habersatter et al., 1998). Fuel oil, fossil gas, petrol, and biofuels all have values ranging between 1.09 and 1.35 MJ.

Table 3. Energy resources used for average Swedish electricity use *(Lundgren, 1992)*

Energy resource	MJ energy resource per MJ electricity
Fossil oil	0.064
Fossil gas	0.0093
Coal	0.040
Peat	0.0045
Biofuels	0.045
Uranium[1]	1.60
Hydropower[2]	0.588
Sum	2.35

1 Calculated as MJ in uranium. 35 % efficiency is used in the conversion of MJ nuclear electricity to MJ in uranium, i.e. 35 % of the theoretical heat obtained in the fission process can be utilised as electricity.

2 Calculated as MJ potential energy. 80 % efficiency is used in the conversion of MJ electricity to MJ potential energy.

Heavy metal contamination of arable land

The seven most commonly measured heavy metals were included in the assessment (Table 4). These metals are those included in the Swedish regulations for heavy metal content in sewage sludge spread on arable land (SFS, 1993; SNFS, 1994). No validation of their respective possible toxicity was performed.

Table 4. Heavy metals included in the environmental assessment.

Pb	Lead
Cd	Cadmium
Hg	Mercury
Cu	Copper
Cr	Chromium
Ni	Nickel
Zn	Zinc

Eutrophication

Eutrophication may have an influence on both terrestrial and aquatic systems. In aquatic systems, the eutrophication may lead to an increased production of biomass. The decomposition of biomass requires oxygen, which may lead to oxygen deficiency. Since the impact is site-specific, different scenarios are presented by Lindfors et al. (1995). In this study the eutrophication impact was calculated following a conservative approach by using the Maximum scenario shown in Table 5. When making case studies, the choice has to be adjusted to conditions valid for each case.

Table 5. Eutrophication weighting factors. *(kg O_2-equivalent per kg compound emitted)*

	N to air	P-limited	N-limited	N-limited + N to air	Maximum
NO_x to air	6	0	0	6	6
NH_3 to air	16	0	0	16	16
NH_4 to water	0	0	15	15	15
NO_3 to water	0	0	4.4	4.4	4.4
P to water	0	140	0	0	140
COD to water (chemical oxygen demand)	0	1	1	1	1

Global warming

Several gases contribute to the greenhouse effect. The gases that have a direct global warming impact are CO_2 of fossil origin, CH_4, and N_2O and these are included in the impact assessment. There are also gases that indirectly contribute to the greenhouse effect; e.g. non-methane hydrocarbons (NMHC), carbon monoxide (CO), and nitrogen oxides (NO_x) but no reliable data concerning their impact were at hand. Other substances like CFCs (freons) were not used in the system under study. The different gases were compared by using Global Warming Potentials (GWPs) weighting factors (Lindfors et al., 1995), see Table 6. The GWPs were expressed as CO_2 equivalents. I have chosen to use a time frame of 100 years, since that covers a period somewhat longer than the perspective of the study, but not too long to be possible to comprehend.

The CO_2 in bio-fuels was not included in the calculations as it is non-fossil, and thus does not contribute to any extra warming impact. This is at least what most scientists claim. In this study, biofuels were a small part of the total energy consumption, so the assumption did not have any large impact on the results.

Table 6. Potential global warming weighting factors. *(g CO_2-equivalent per g compound emitted)*

Compound	20 years	100 years	500 years
CO_2-fossil	1	1	1
CH_4	62	24.5	7.5
N_2O	290	320	180

Acidification

The acidification impact was calculated using the approach suggested by Finnveden et al. (1992), where the effect is defined as the amount of protons released in terrestrial systems. Two scenarios are given. One that takes emission of nitrogen compounds into account (maximum scenario), and one that does not (minimum scenario). The reason is that the nitrogen has to be leached out from the system to have an acidifying impact. In typical situations in Europe, some 10 – 30 % are leached. Thus, in practice, acidification should be somewhere in between the two scenarios given. In this study, it was primarily the maximum scenario which was used.

Table 7. Acidification weighting factors. *(mol H^+ per g compound emitted)*

Substance	Minimum scenario	Maximum scenario
SO_2	0.031	0.031
HCl	0.027	0.027
NO_x	0	0.022
NH_3	0	0.059

Photo-oxidant formation

When reporting photo-oxidant formation, only ozone production is usually considered. There are however other oxidants of environmental interest, but they have been less studied, according to Nordic Guidelines on Life-Cycle Assessment (Lindfors et al., 1995). Following the recommendation in that publication, two different sub-categories were reported here – VOCs (Volatile Organic Compounds) aggregated according the POCP (Photochemical Ozone Creation Potentials) concept, and nitrous oxides aggregated as NO_x (NO_2). The POCPs were expressed as ethene-equivalents (Table 8). The scenario suggested by Heijungs et al., which uses average peak ozone data from three trajectories, was used. It was complemented with the POCP-factor for carbon monoxide (CO) given by Finnveden et al. (1992) (ordinary Swedish background during 0-4 days).

Table 8. Photochemical ozone creation potentials (POCP). *(g ethene-equivalents per g emitted compound)*

Compound	POCP
CH_4 [1]	0.007
VOC (Volatile organic compounds) [1]	0.337
NMHC, Non methane hydrocarbons (VOC-CH4) [1]	0.416
CO, ordinary Swedish background during 0-4 days [2]	0.040
Ethanol [1]	0.268

[1] Heijungs et al., 1992

2 Finnveden et al., 1992

Impact categories excluded

The impact categories excluded might be important as well, but some of them are not affected much by the different system choices (e.g. depletion of stratospheric ozone), and some categories are not yet scientifically well documented and agreed upon (e.g. ecotoxicological impacts).

Validation

Validation of the whole-systems results is almost impossible without construction of the real systems. Since the purpose in using modelling and simulation is to test different systems without (or before) building them in practice, this was not an option. Instead the performance of each sub-model was validated for different conditions and validity ranges. If all sub-models are validated and run under conditions within the validity range, then the entire system put together can be said to be validated.

Nomenclature

energy consumption	though energy cannot be consumed (or produced) in a thermodynamical sense – just transformed to another quality – I will use this term as it is commonly used and in an easy way describes the delivery of energy to and from the systems
energy production	see energy consumption
scenario	(here) a model of a complete system, consisting of several sub-models of sources, transport, treatments, processes etc.
blackwater	part of the household wastewater coming from toilets
greywater	part of the household wastewater coming from baths and showers, dish- and laundry-washing
household waste	here defined as easily degradable wastes originating from foodstuffs handled in households
organic waste	here defined as easily degradable waste originating from household wastes, sewage water, and sometimes manure
manure	here always used in the meaning animal manure
cyclic system	reconnected systems, that feed back e.g. organic waste
food-system	here the entire system producing food, i.e. including organic waste management and other fertiliser production

Organic waste management system

This chapter presents the study of organic waste management and agricultural grain production. It starts with methodological issues and model descriptions and then continues with results and discussion.

Society produces several different kinds of easily-degradable organic wastes (here called organic wastes). Examples are park and garden waste, leftovers and refuse from groceries and restaurants, grease separated from sewage in restaurants and food industries, wastewater from households and food industries and waste products from food industries. The latter are, however, largely utilised as animal fodder (Brolin et al., 1996). The largest proportion of all organic wastes is manure from animals (if harvest residues left on fields are not taken into account).

The treatment of organic waste can consist of biological degradation methods such as composting and anaerobic digestion, of incineration (which is often energy-demanding due to large water content), of pyrolysis, or of landfilling. Wastewater is often treated using combinations of biological and chemical methods, followed by (or combined with) sludge separation. The sludge can then be treated by the methods mentioned above, the effluent water is often let out to surface watercourses. Biologically treated wastewater might also be stored and used for irrigation of agricultural crops. Alternative treatments of wastewater are source separation (i.e. no mixing of different fractions) or initial sludge separation, followed by sludge treatment as above and simpler treatment or polishing of the water fraction. A broad spectrum of polishing methods is at hand. Examples are infiltration in soil or constructed filter beds, vegetative filters, and wetlands.

System boundaries

Since the focus in this study was the interaction between housing areas and agriculture, only household (easily degradable organic) wastes, household sewage water and animal manure were included. However, other kinds of organic waste could easily be included in later studies. Treatments chosen to be studied here were current standard methods in Sweden and methods aiming at resource (plant nutrients, organic matter, and/or water) utilisation. Landfilling of organic waste will soon be prohibited and has been shown to be a practice which is not recommendable from an environmental point of view (Sonesson et al., 1997) and was thus not included. Incineration was not included since the burning of organic waste, to be practically possible, must be performed in mixed-waste incineration. The energy is utilised, although doubts about the energy value of watery waste can be raised, but the organic matter is lost. The plant nutrients are also lost since it is not likely to be possible to spread ashes on arable land due to their content of environmentally harmful substances. Furthermore, since the focus was on small-scale settlements, no conventional sewage plant was included but can also possibly be included in later studies. The parts included and the system boundary are shown in Figure 7.

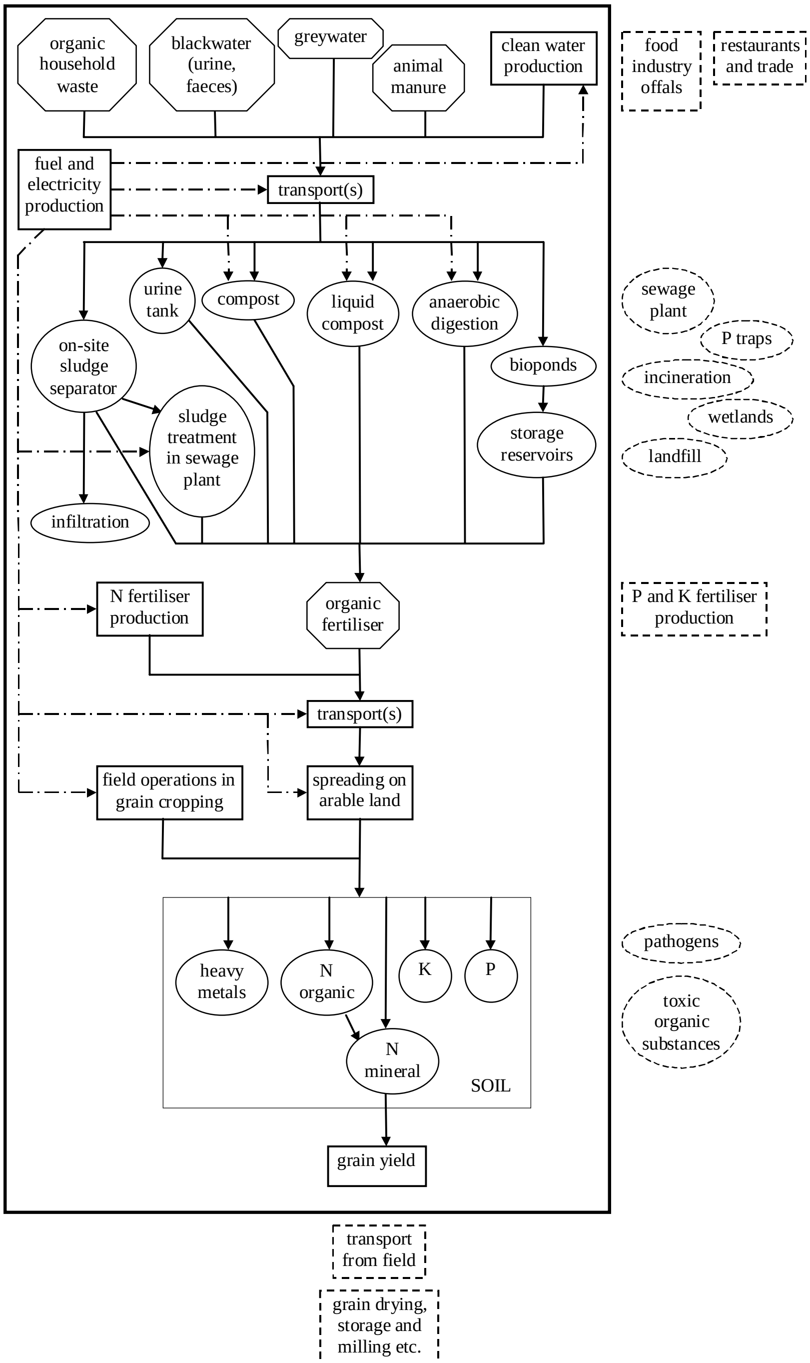

Figure 7. Parts included and system boundary in the organic waste management study.

The housing area which served as the pattern for the study was a rural settlement with 32 households (100 inhabitants) in single-family houses and one agricultural farm where all residues and manure were spread. No industries or food processing plants were taken into account, since the focus in this part of the work was on the connection between households and agriculture. The transport distances were short. All residue transport distances were set to two kilometres, i.e. all treatments were situated in or nearby the village. The same was valid for the transport from treatment to the field for spreading. The only exception was the distance for transportation of on-site separated sludge to sewage plant and the treated residues returned, which was 30 km. Many of the in-data were average figures valid for “normal” households and treatment plants in Sweden, and some were site-specific data from the "pattern" settlement. Thus, the study should be valid for similar locations in the southern and central parts of Sweden.

The agricultural parts included comprised the fate of nitrogen in soil, cereal field grain production, energy consumption and air emissions for grain production field operations, energy consumption and air and water emissions from possible additional N-fertiliser production, and manure from 15 dairy cows including young-stock. The number of animals was chosen to roughly represent the number of animal equivalents[1] per person in Sweden (Naturvårdsverket, 1989; Statistics Sweden, 1997). Since the simulated system had 100 individuals that made 0.15 animal equivalent per person. All other parts of the agricultural system were excluded since they were assumed not to change between scenarios. In drawing conclusions about the entire food system, this is of course a weakness of the study, since the agricultural parts not included are likely to have large environmental impacts.

In order to get the scenarios to produce equal amounts of grain, extra mineral nitrogen fertiliser was added when needed. The need was calculated as the difference between the scenario producing the largest amount of grain (Scenario 0 in Figure 8) and the actual scenario. This was accounted for as “fill-up N-fertiliser production” (Scenario 1 in Figure 8). In some cases, where results for household residues only were presented, the acreage could not produce enough grain, although mineral fertiliser was added (Scenario 2 in Figure 8). The reason was the large difference in acreage utilised for the spreading of organic fertilisers when household residues alone were considered. To compensate for this deficiency, additional acreage had to be included. This acreage was conventionally cropped using mineral fertilisers. When this addition was necessary, the extra nitrogen fertiliser and the extra acreage were accounted for as “fill-up grain production”.

[1] Animal equivalents as given by the Swedish EPA concerning “Environmental protection in animal husbandry” (Naturvårdsverket, 1989). One animal equivalent represents one full-grown cattle, three sows, ten growing-fattening pigs, etc.

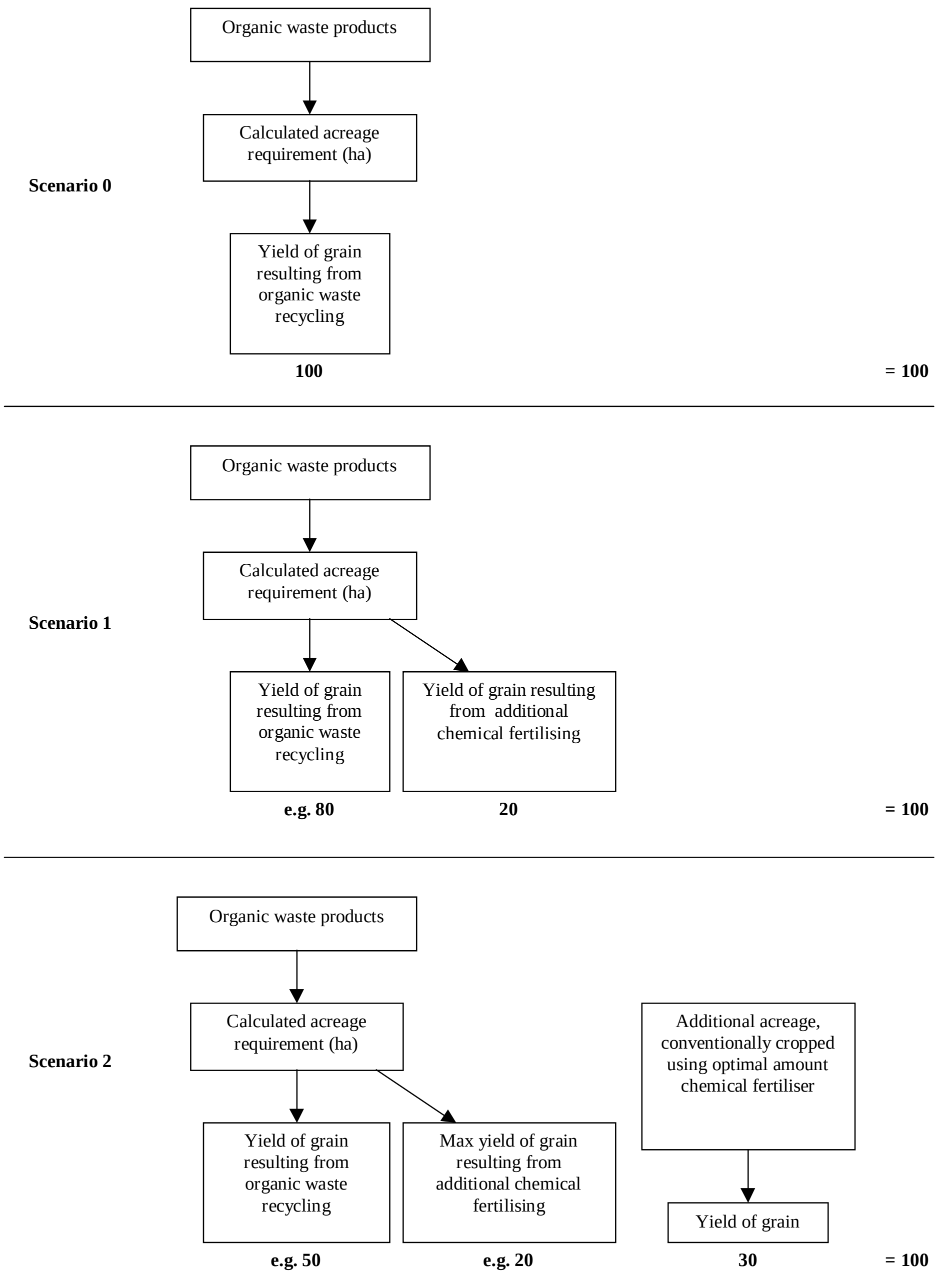

Figure 8. Different scenarios of acreage and chemical fertiliser requirements to obtain equal production of grain in the different organic waste management scenarios. Scenario 0 = the most grain-producing org. waste scenario; Scenario 1 = the org. waste scenarios where additional chemical nitrogen fertilising (on the acreage used for org. waste spreading) is able to rise the yield of grain to the level of Scenario 0; Scenario 2 = the org. waste scenarios where both optimal additional chemical nitrogen fertilising (on the acreage used for org. waste spreading) and additional conventionally cropped acreage is needed to produce as large yield of grain as Scenario 0 does.

Production and construction of equipment, vehicles, infrastructure, etc. were not taken into account, i.e. only resources and energy consumed within the system were accounted for. The same applied to air and water emissions. This may have been a serious simplification of the problem, thus making the results less representative for the real system performances. However, air emissions occurring in the production of chemical nitrogen fertilisers, fuels and electricity were included.

Assumptions

Electricity was assumed to be produced as Swedish average electricity use, i.e. about equal parts of hydropower and nuclear power. Minor contributions come from biofuels, wind, oil, and coal. Implications of the use of electricity produced in oil-fired power plant are described in the chapter "Sensitivity analyses; organic waste management".

All local transport (all transport except that to and from the sewage plant) was 2 km one-way. The scenarios were also run with these transport distances set to 10 km in a sensitivity analysis presented in the chapter "Sensitivity analyses; organic waste management".

Data for energy consumption and emissions in nitrogen fertiliser production (including transport) are average figures valid for Germany, published by Patyk (1996).

Scenarios

Twelve different organic waste treatment scenarios were simulated in this study. Six different main treatments were each combined with both non-separating toilets as well as with urine-separating toilets.

All scenarios (except scenarios 5 and 6) had sludge separators followed by infiltration for treatment of either all of the wastewater (scenarios 1-4) or of the greywater (scenarios 7-12). The sludge was spread on arable land or included in the treatments in all scenarios. The scenarios 7-12 had blackwater separating sewage systems where grey- and blackwater were kept apart.

The twelve scenarios were:

1) *SludgeToSewPlant*, digestion of sludge from single household sludge separators in sewage plant, household waste was home-composted and recycled to arable land. Manure was stored, transported and spread independently.

2) *SludgeToSewPlantUrineSep*, as 1) but human urine was separated in the toilets and stored, transported and spread independently.

3) *SludgeToAgr*, spreading of sludge from single household sludge separators on arable land without any treatment, household waste was home-composted and recycled to arable land. Manure was stored, transported and spread independently.

4) *SludgeToAgrUrineSep*, as 3) but human urine was separated in the toilets and stored, transported and spread independently.

5) *Bioponds*, mechanical and biological treatments of sewage water (including household wastes) in open-air bioponds followed by storage reservoirs. Manure was stored, transported and spread independently.

6) *BiopondUrineSep*, as 5) but human urine was separated in the toilets and stored, transported and spread independently.

7) *CompostToilet*, composting of faeces, urine and household wastes. Toilets with dry handling of the excrements. Manure was stored, transported and spread independently.

8) *CompostUrineSep*, human urine was separated in the toilet and stored, transported and spread independently. Faeces were separated from flush-water and composted together with household wastes. Manure was stored, transported and spread independently.

9) *LiquidCompost*, liquid composting of blackwater, household wastes and manure.

10) *LiquidCompostUrineSep*, as 9) but human urine was separated in the toilets and stored, transported and spread independently.

11) *AnaerobDigestion*, anaerobic digestion of blackwater, household wastes and manure.

12) *AnaerobDigestionUrineSep*, as 11) but human urine was separated in the toilets and stored, transported and spread independently.

The anaerobic digestion scenarios were simulated in two different modes – with or without hygienisation (heating to 70º C) of incoming material, making fourteen scenarios in all for the energy calculations.

The scenarios 5, 6 and 9-12 included kitchen disposers to macerate and blend the household waste with the sewage water.

Models

The general structure of the organic waste management model, exemplified by the anaerobic digestion with urine separation scenario (AnaerobDigUSep), is shown in Figure 9. At the top, waste sources are initiated. Then follows the sub-models for calculation of energy turnover and emissions in treatments, transport and spreading of the wastes. Finally the soil nitrogen turnover, grain production, and field-operation energy consumption are calculated in the agriculture sub-model. This block also contains the initialisation of manure. The sub-models are described in more detail below.

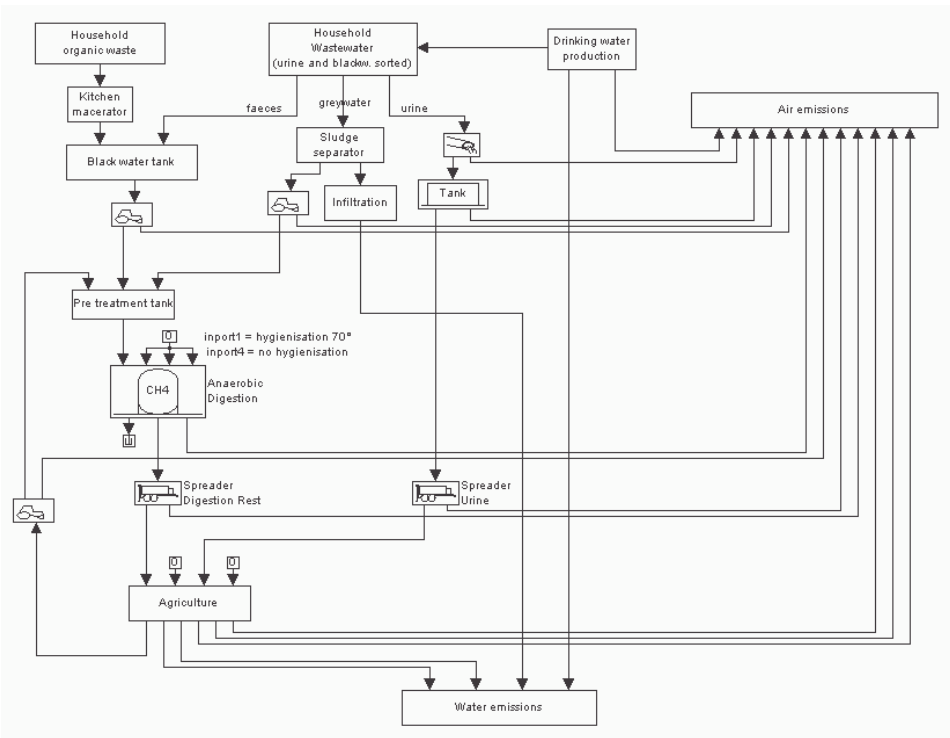

Figure 9. General structure of the organic waste management model. The anaerobic digestion with urine separation scenario.

Figure 9 shows the most complicated system simulated in this study. An example of a simpler system, the scenario where sludge from on-site sludge separated wastewater was transported straight to agricultural land, is shown in Figure 10.

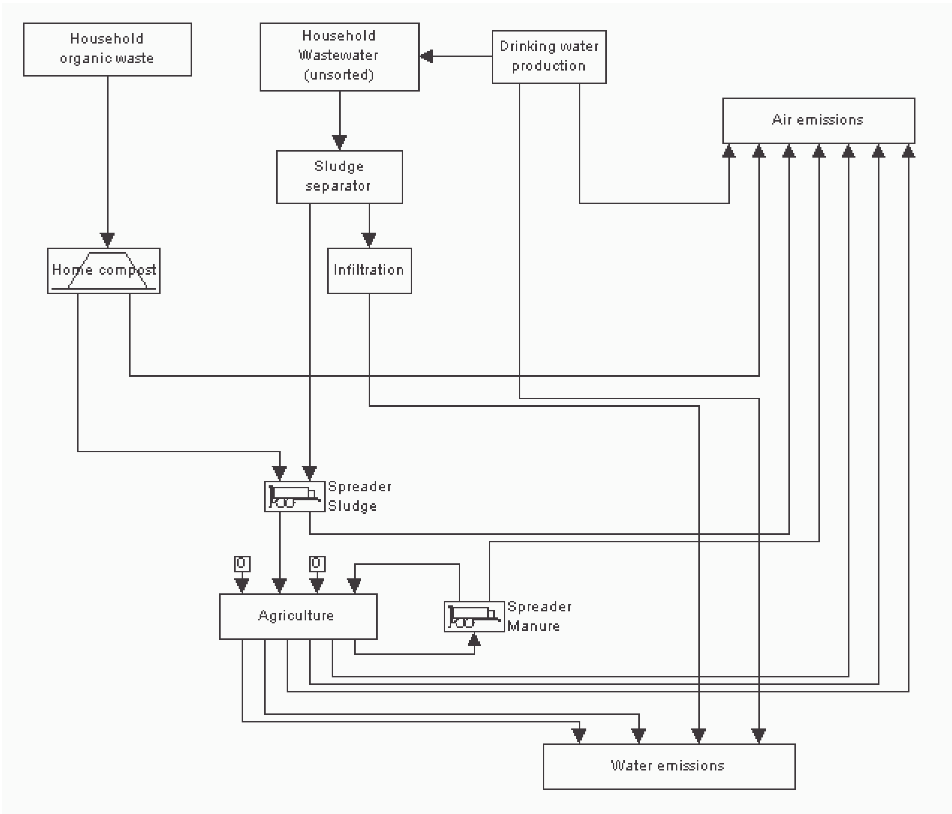

Figure 10. The "SludgeToAgr" scenario.

Sources

The sources, or input vectors, were wastewater and household wastes from 32 households and manure from 15 dairy cows and their young stock.

The wastewater was defined as urine, faeces, flush water including paper, and greywater (Figure 11). These were identical in all scenarios except for the amount of flush water which varied depending on whether or not there was urine separation (see “Urine separation” below). The household sources have been thoroughly described in Nybrant et al. (1996). Some minor modifications were made following the Swedish Environmental Protection Agency “template” figures for content of nutrients and heavy metals in Swedish household wastewater (1995). It was assumed that 75 percent of the total blackwater was produced at home and 25 percent at work. The latter part was not taken into account. Concerning the household data, these were yearly average amounts and/or average content data assumed to be valid for “normal” households in Sweden. In reality, the variation could be large due to differences in e.g. age, gender and diets, but that was not possible to take into consideration in this study. In addition, no sufficiently detailed data were available.

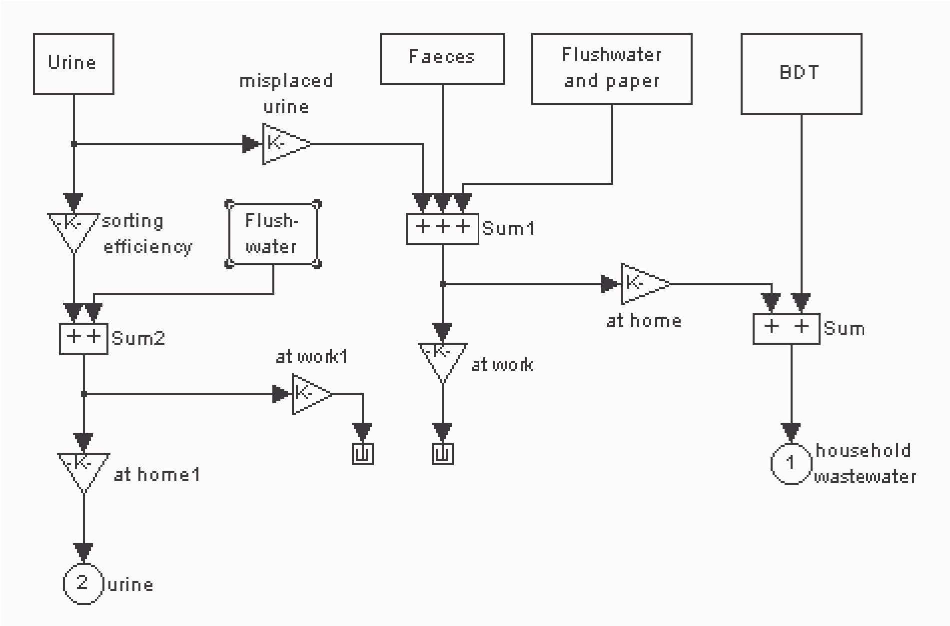

Figure 11. The household wastewater (with urine separation) initialisation block.

In practical urine separation, the urine is sorted away “at source” (i.e. never mixed with the faeces) in two-bowled toilets. The separation efficiency is assumed to be 80 percent, meaning that 20 percent of the urine is misplaced and thus added to the faeces fraction (Jönsson, 1997). The urine is then transported in a pipe to a storage tank where it is stored for several months. It is collected, transported and spread on arable land with a tractor-drawn liquid manure spreader. In the models, the separation is carried out by not mixing the input flows. The only difference in incoming values is the amount of water. For ordinary toilets, it was assumed that each toilet flushing consists of 6 litres and that each person flushed 5 times per day, making 30 litres per person and day. When urine separation was used, we assumed two big flushings of 6 litres and 5 small flushings for the urine-bowl of 0.2 litres, i.e. 13 litres per person and day.

The manure sub-model is an input vector that defines the amount and content of manure from fifteen milking cows including young stock. The content data used in this study came from a few measurements on liquid manure delivered to an anaerobic digestion plant (Johansson, 1997b). Thus losses of nutrients in the animal house, storage and transport were included but the emissions due to them were not accounted for. The content probably varies a lot depending on feeding and production strategies. This should be further examined when real case studies are performed. The amount of manure was calculated from Claesson & Steineck (1991). The manure was handled as a slurry in all scenarios.

Clean water production

Water of drinking quality is used in households for different purposes. In the production of this water, chemicals are added to achieve hygienic and functional demands. The energy consumption for production and transport of these chemicals is calculated in the model (Figure 12). In addition, energy consumed in pumping and in transport in connection with service work is also calculated. Air emissions from chemical production and all transport, as well as water emissions from the chemical production, are calculated. This model was developed by Kärrman (1999). Data were taken from Tillman et al. (1996) and The Swedish Water & Wastewater Association (1994). Vinnerås (pers. comm) contributed the service transport part (unpublished data from an investigation of the fresh-water production in Stockholm made in 1998).

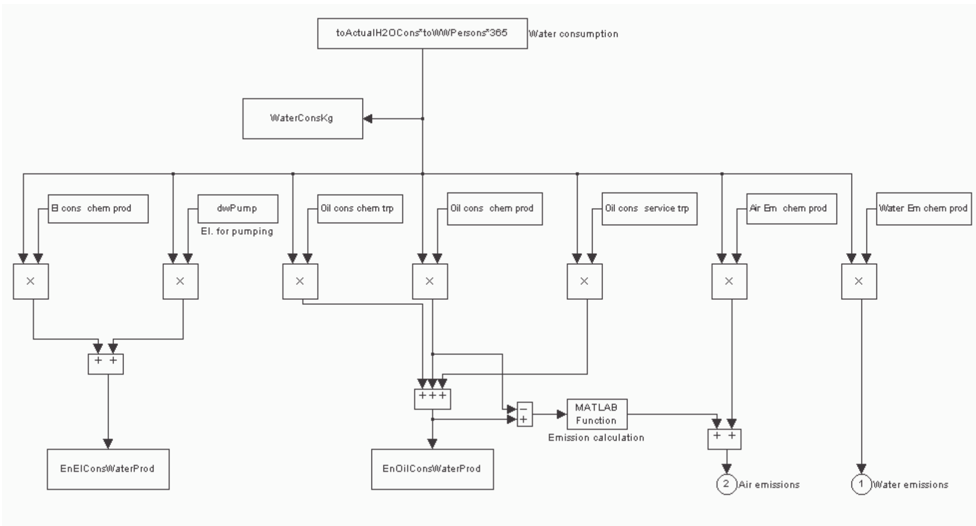

Figure 12. Clean-water production sub-model.

Treatments

Very few data seem to be available on the *sludge separator* process. Those that are on hand are disparate, probably due to large variations in the functioning of different types of separators and large variations in incoming sewage water content. The sludge separator sub-model is shown in Figure 13.

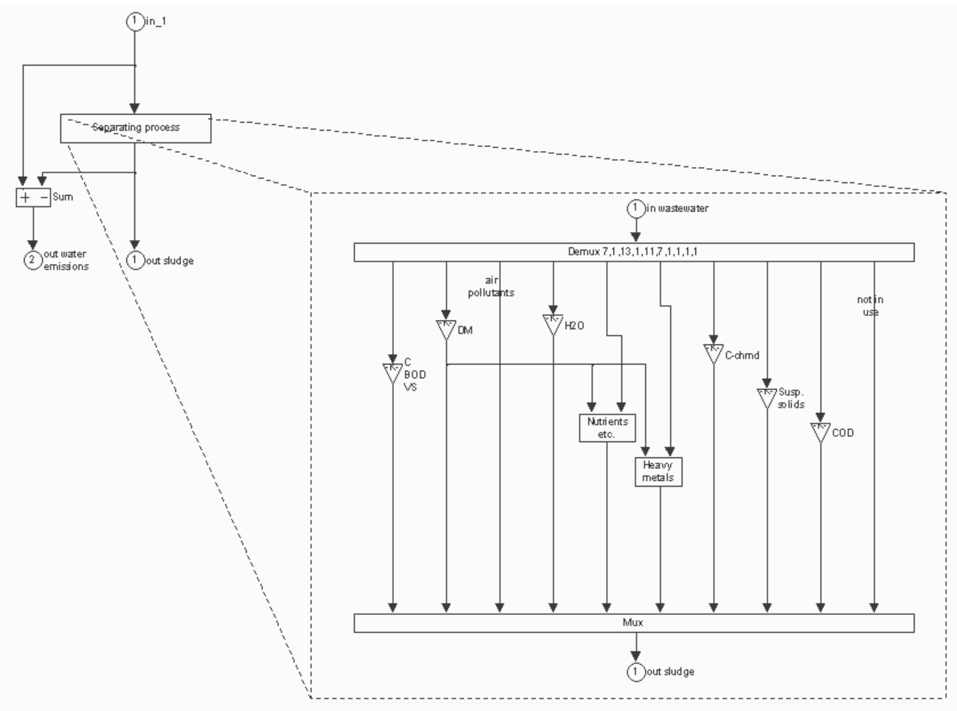

Figure 13. The sludge separation sub-model. Upper level at left, lower level at right.

I have chosen to use the approximate cleaning efficiencies for organic material (BOD_7/COD, 10 percent reduction) and suspended solids (70 percent reduction) given by the Swedish Environmental Protection Agency (1987). The proportions of nutrients and heavy metals sedimented are calculated from measured contents in on-site sludge separators (Andersson, 1992; Landers, 1995) and contents in incoming household sewage water (Sundberg, 1995) simply by multiplication of the dry matter content in the wastewater by the chosen content per kg dry matter of nutrients and wastewater, i.e. output from the model is sludge with a fixed content of plant nutrients and heavy metals. The model is constructed so that the total amount of a substance in this predetermined sludge could never be larger than the amount in incoming wastewater. If the in-data produce such a result, the model set the output equal to the amount in the wastewater, i.e. all of the substance ends up in the sludge. This is of course not the case in reality, but with the data used in the study this will occur for Cu, which will be further discussed when the results are presented. For more details see Appendix 1.

Neither the structure of the model nor the data used is very good, but for the purposes of this study, I judge the model to be sufficiently appropriate. However, if these models are to be used in case studies where the aim is to evaluate the performance of sludge separators in detail, the model should be further elaborated. At the very least, the input data have to be adjusted to local conditions.

The model was not modified when used for urine-sorted wastewater or for greywater. One analysis made on sludge from an on-site sludge separator fed by urine-sorted household wastewater indicates that the performance is not changed (Johansson, 1997a).

The outlet water was infiltrated in an *infiltration bed.* As for the sludge separator, I have chosen to use the cleaning efficiencies given by the Swedish Environmental Protection Agency (1987). They are quite crude but appropriate since the focus in this study was not on water emissions and since different infiltration structures have a wide range of performance. This sub-model is shown in Figure 14.

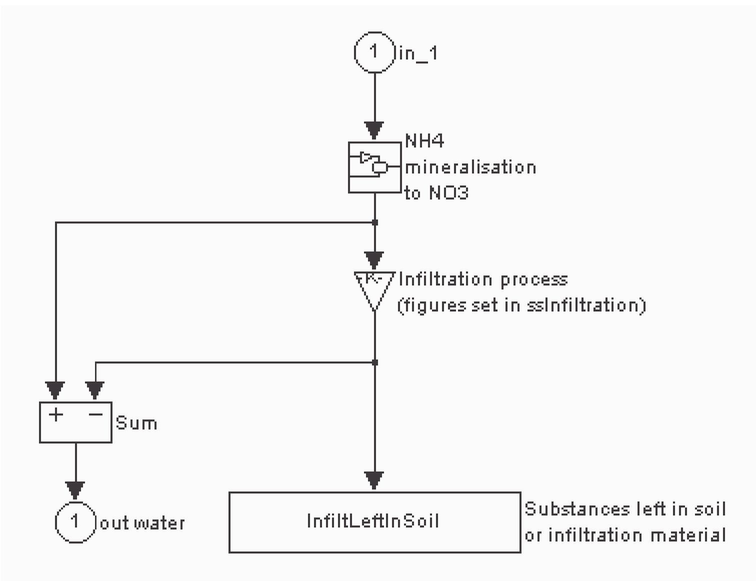

Figure 14. The infiltration sub-model.

When the *sludge is transported to a municipal sewage plant,* it is put in at the dewatering stage prior to the digestion chamber. It is digested together with the raw sludge from the sewage plant treatment of municipal wastewater. The sludge from "our" single houses was a very small part of the total sludge digested. The model outputs were the differences between two parallel sewage plant sub-models, one with municipal wastewater only, and the other with the same municipal wastewater plus the sludge from the single household sludge separators. The reason for this model construction was that the model is validated for large-scale treatment of mixed sewage water only. The sewage plant sub-model is described in detail in Dalemo (1996).

At present, about 50 % of the digested sludge from many sewage plants in Sweden has a too large a content of heavy metals or organic pollutants to be spread on arable land. In practice, this fraction is put on landfill or incinerated. I have chosen to put it on *landfill*, since that is the most common method of treatment. The landfill process is divided into two lifetime periods, a short surveyable time and an infinite remaining time. The surveyable time for organic wastes is defined as the time it takes for the process to slow down to a pseudo-steady-state. About 100 years is required to reach this stage. Only the emissions in this period are accounted for in the results. The model is described in detail in Mingarini (1996) and Björklund (1998).

When the *sludge is transported straight to arable land* for immediate spreading, this is done without any further treatment. In practice, there might be a hygienisation treatment before spreading, such as e.g. liming or extended storage, but I have not found any data concerning degradation of the material produced by such treatments. The major effect is probably that some, or even much, of the nitrogen is lost. It was assumed that the sludge separators were emptied once a year, thus did not produce any large amounts of e.g. methane. If the separators are emptied more often, there are data indicating that more nitrogen could be left in the sludge (Carlsson, 1995).

The *bioponds* are one or two constructed open-air ponds where the biological activity is high. Most of the organic material is decomposed and some particles are sedimented (Gotlands kommun, 1996). The World Health Organization (WHO) recognises such "waste stabilising ponds" as a good method for hygienisation of wastewater (Bengtsson et al., 1997). It is further mentioned that it is a cheap, robust and simple method of wastewater treatment. The only disadvantage is the relatively large area of land required. The retention time can be between 10 to 30 days (Larsson, 1995) or even up to 50 days depending on the design and effluent quality required (Mara & Cairncross, 1989). The sewage water is transported in a pipe system to the ponds without any preceding sludge separation.

From the bioponds, the effluent water might be discharged into surface watercourses, directly used for irrigation, or pumped to maturation and storage reservoirs for use as irrigation water during the next irrigation season. In this study, it was assumed that the water was stored in such reservoirs and used for irrigation. For hygienisation purposes, the minimum storage time was set to half a year (valid in Sweden). The reservoirs should hold one year of wastewater production so that all water can be used for irrigation and nothing let out to recipient waters. During the irrigation season there might be a need for several parallel storage reservoirs in order not to re-infect the cleansed water. Mara & Pearson (1999) describe a hybrid system comprising a series of bioponds followed by a single storage reservoir. The idea is that during the non-irrigation season, the biopond effluent is discharged into the storage reservoir, while during the irrigation season, the effluent is used directly for restricted or unrestricted irrigation, depending on its hygienic standard. This system should be simpler to operate and, above all, it reduces the land area required.

In this study, it was assumed that the water was distributed to arable land in a high-pressure pipe system. Sprinklers were used for the irrigation. Electric energy consumption for pumping was calculated in separate sub-models (not shown).

The biopond sub-model is shown in Figure 15. The storage reservoir model is similar to the biopond model. They calculate the degradation of organic matter and the losses of nitrogen and phosphorus. Data used for the calculations were measured content of total nitrogen, phosphorus and organic matter (biological and chemical oxygen demand) in incoming sewage water and in outgoing water from both the bioponds and the storage reservoirs in three biopond-systems on Gotland, Sweden (Gotlands kommun, 1996). Average degradation rates were 95 % for BOD_7, 88 % for COD, 80 % for N-total and 76 % for P.

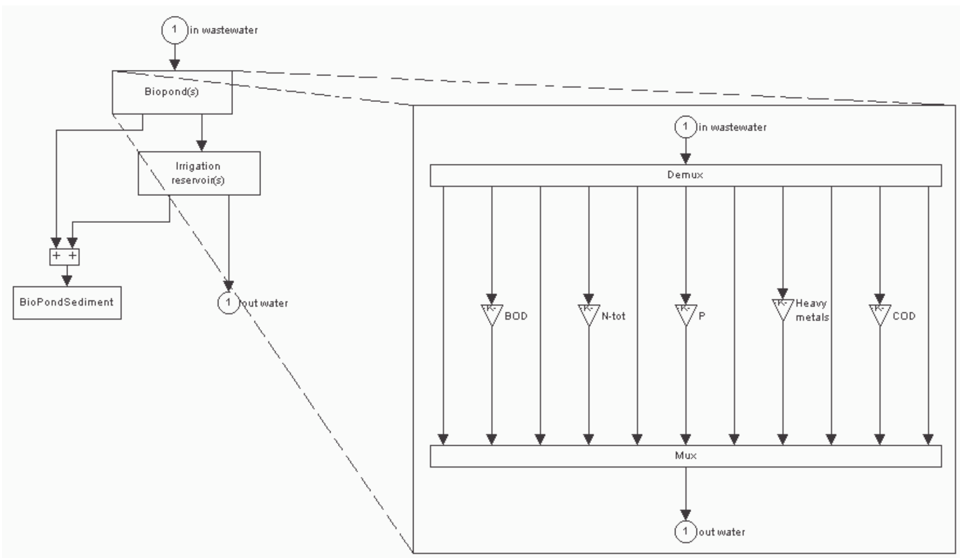

Figure 15. The biopond sub-model. Upper level at left, lower level at right.

The *compost* process sub-model is described in detail in Sonesson (1996). The compost is assumed to be “well-handled”, i.e. almost no anaerobic conditions occur. The material is given time to mature. The sub-model includes options for mixing and aeration in outdoor windrows by a mobile mixing machine and for cleaning of the outlet gases in a biofilter which traps almost all nitrogen but neither of these were used in this study.

The *liquid compost* sub-model was constructed in this study (Figure 16). Data concerning reactor, process, energy consumption, and air emissions were taken from Skjelhaugen (1998), Sæther (1997), and Norin (1996).

Air is pumped in at the bottom of the reactor to assure that aerobic conditions are maintained. The air bubbling up also creates a slow blending of the slurry. Process temperature is 55-60 º C. Electricity is used for air pumping, for pumping of the liquid, and for foam cutting. The total energy consumption is calculated in the model. The outlet gases are used for heat-exchanging the inflow of air. In this cooling of the outlet gases, some moisture is condensed and fed back to the reactor. The outlet gases are then supposed to be cleaned in a biofilter (here peat). Also here, excessive water is fed back to the reactor. The filter material was assumed to be recycled to arable land together with the slurry, thus very little nutrients were lost. In the model, the post treatment storage period is assumed to be six months (26 weeks). Skjelhaugen and Donantoni (1998) give reductions of VS, DM, COD, N-total and N-NH_4 after 45 weeks (10 month) of post treatment storage. I assumed that the losses were linear over time and thus multiplied their numbers by 26/45. This gave 12 % losses of ammonia and 9 % of total nitrogen. The microbial turnover in the slurry is probably not linear over time as assumed here, so this approximation might be a source of erratic results. For more details see Appendix 2.

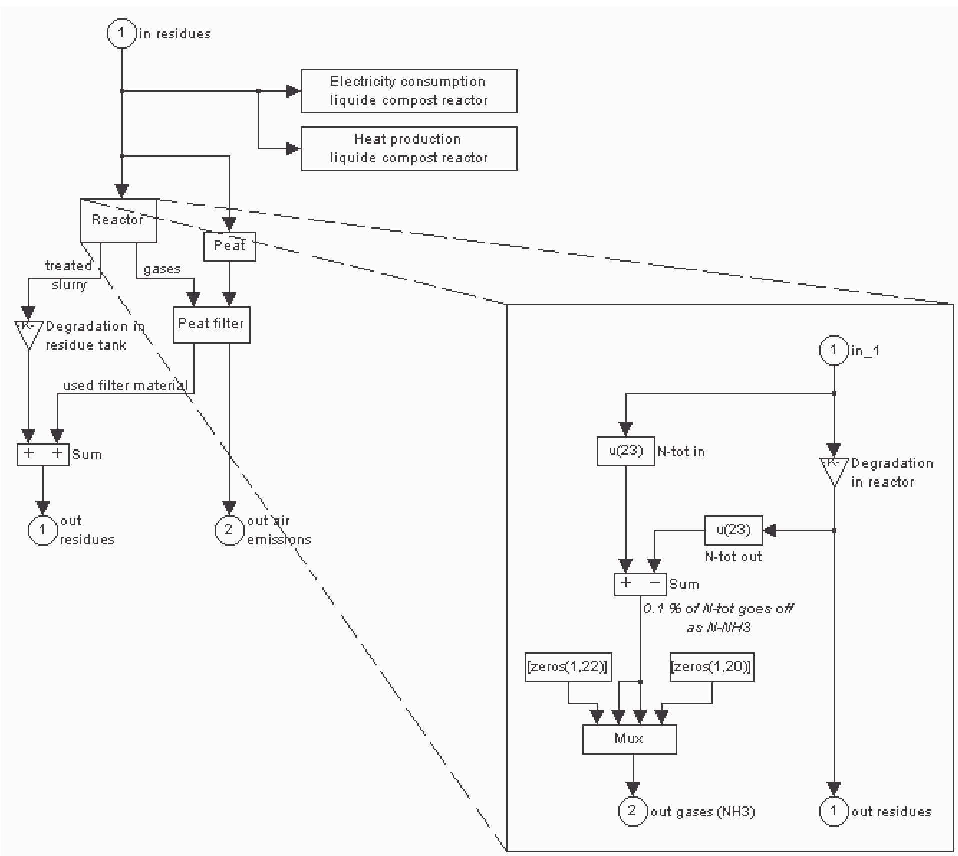

Figure 16. The liquid compost sub-model.

The *anaerobic digestion* sub-model describes a continuous, single-stage, mixed-tank digestion chamber operating under mesophilic conditions (37°C). The incoming material can, if chosen, be hygienised (heated to 70^{O} C) or sterilised (heated to 130^{O} C). In this study, the process was simulated both with and without hygienisation. Sterilisation was not used here. The loss of ammonia during six months post treatment storage was assumed to be 1 percent. The model is thoroughly described in Dalemo (1996).

In the scenarios with urine-separating toilets, the *urine collection* is described in two sub-models – urine pipe and urine collection tank (Jönsson et al., 1999). These are identical except for the optional possibility of including in-leakage to the pipe (Figure 17). Both calculate the transmission of urea to ammonia and the emission of kg N in ammonia to air. The ammonia emission is assumed small and thus not affect the amount of nitrogen in the solution. The in-data are pH and temperature of the urine, the ratio between volume of the exchanged gas and the liquid, the ratio in the gas between actual ammonia pressure and saturation pressure, the liquid volume and finally the amount of nitrogen in ammonia per volume. The calculations are based on Svensson (1993).

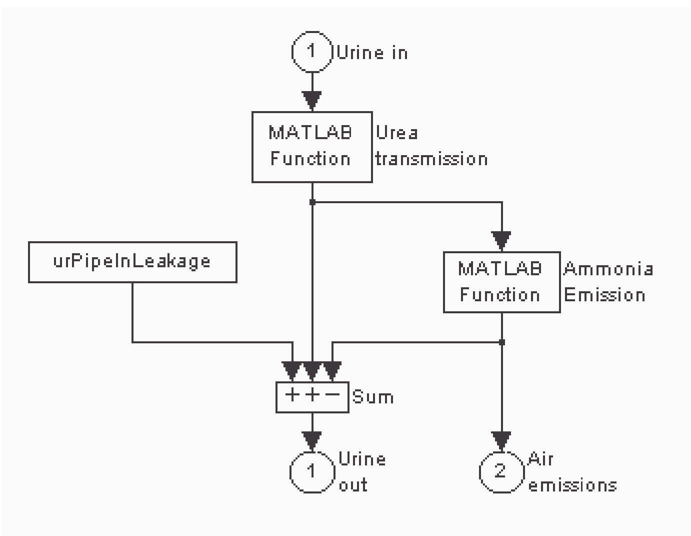

Figure 17. The urine-pipe sub-model.

The *kitchen macerator* (disposer) makes it possible to mix and transport the household wastes together with the sewage water in the sewage system. The sub-model calculates the energy consumption, 3.7 MJ/person and year, and additional water consumption, 3.4 l/person and day (Nilsson et al., 1990). It was used in the biopond, liquid compost and anaerobic digestion scenarios. The model was developed within this study (Figure 18).

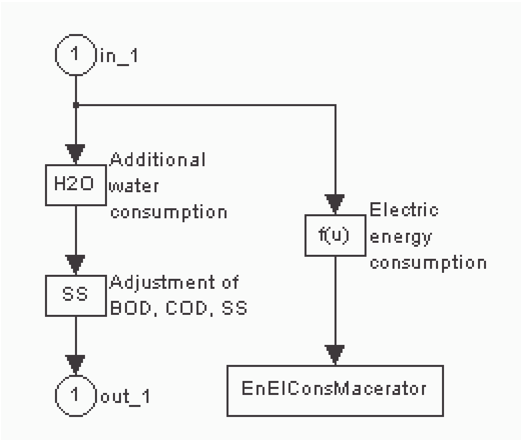

Figure 18. The kitchen macerator sub-model.

Transport

Two different types of transportation by vehicle were used in the organic waste management study – truck and tractor with manure spreader. The truck was used for the longer distance transportation of the on-site separated sludge to the sewage plant and for taking the digested sludge from the sewage plant to the landfill and to arable land. A tractor with manure spreader was used for all short-distance transportation and spreading of blackwater, urine, sludge, manure, and treatment residues. The vehicle models calculate energy consumption and emissions to air due to the combustion of fuel. All vehicles in the organic waste management study were diesel-fuelled. Emission data are taken from Egebäck & Grägg (1988), Grägg (1990, 1992) and Egebäck & Hedbom (1991) and are the same for all transport.

The *truck* sub-model (Figure 19) is presented in detail in Sonesson (1996). The fuel consumption used here was 4.0 litres/10 km (14.2 MJ/km) for transportation of digested sludge to arable land and landfill. For the collection and transportation of on-site separated sludge, I assumed 4.5 litres/10 km (16.0 MJ/km) since the pumping of sludge is energy demanding.

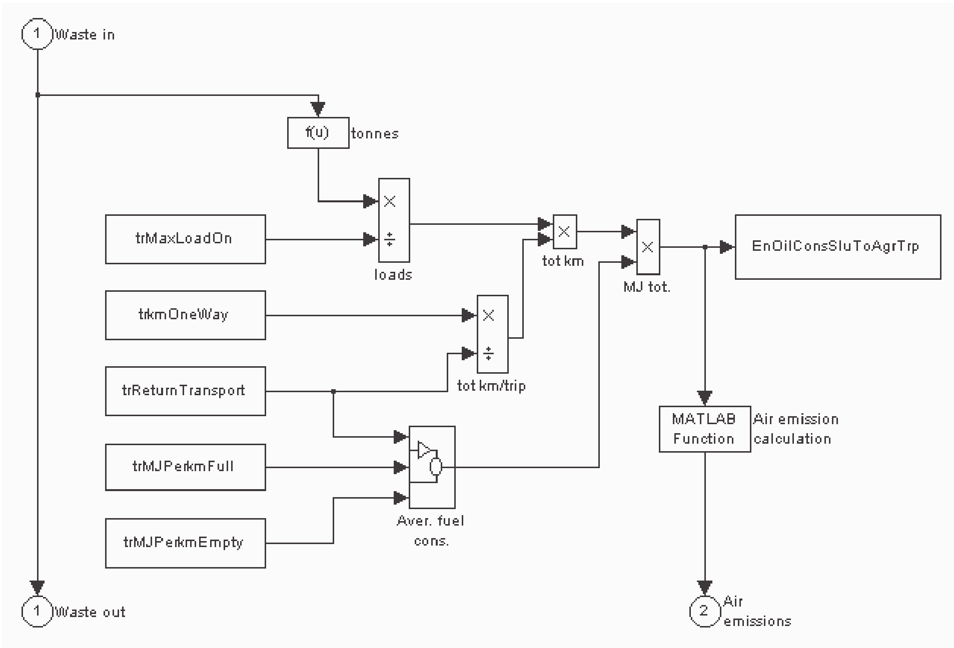

Figure 19. The truck sub-model.

Spreading on arable land

The *spreading of liquid and solid residues on arable land* sub-model (Figure 20) was developed by Håkan Jönsson and later modified by Ulf Sonesson and myself, all at the Department of Agricultural Engineering at SLU in Uppsala, Sweden. The model exists in two versions, which are identical except for the loading-energy demand calculation. The liquid spreader is pump-loaded and the solid materials spreader is loaded by wheel-loader. These models calculate the energy consumption for loading, transport to field and spreading on field; the emissions to air due to the combustion of fuel; hectares needed for spreading; and travel distance on fields. The time for all operations is calculated for use in an economic evaluation modelled in the original version. However, neither economy nor time use were included in this study.

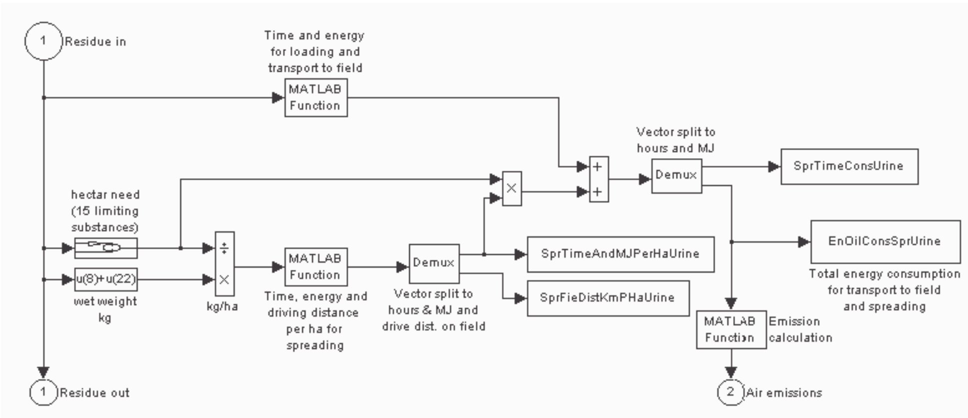

Figure 20. The liquid manure spreader sub-model.

The energy demand is calculated as the mechanical work required to move a certain weight a certain distance with correction for rolling resistance coefficients and efficiencies in engines, pumps and gearboxes. Formulae are taken from Elinder & Falk (1983) and Jonsson (1993). Input variables for the liquid spreader sub-model are: pump capacity; lift height (for pumping the liquid from storage into the spreader); amount residue (wet weight); travel distance from storage to field; spreader tank volume; tractor and spreader combination empty weight; transport velocity; and additional time per load (entering and leaving the tractor etc.). For the solid material spreader, the input variables concerning loading are energy and time consumption for a wheel-loader (VOLVO BM, 1995), all the others are the same as for the liquid spreader model. The energy consumption obtained in the model was judged to be appropriate. Concerning the air emissions, these are calculated using emission data valid for truck engines, which could be a source of error since the emissions vary a lot between different working conditions for tractor engines according to a preliminary study by Hansson et al. (1998). However, no sufficiently detailed data are available as yet.

For *road transport* with tractor-and-manure-spreader, stripped versions of the liquid and solid material spreader sub-models were used. In this study, driving speed for road transport was set to 20 km/h, and one-way transport distance was set to 2 km. For more details see Appendix 3.

Acreage requirement

The acreage requirement in hectares (ha) is calculated from permissible or desirable amounts of 15 substances (N, P, K, S, heavy metals and organic pollutants), spread per ha and year as described in Sonesson (1998). The result is both an output and used internally for the field operation calculations. The individual organic wastes were spread on separate acreage, implying that the number of hectares and grain production per ha is different between scenarios.

Agriculture

The agriculture sub-model (Figure 21) comprised two separate sub-sub-models – nitrogen soil turnover (Soil) and grain production (Crop). To be able to evaluate the contribution from different organic wastes, four parallel Soil-Crop-combinations were included. Furthermore, the field-operations energy consumption is calculated, and the amount and content of manure is initiated.

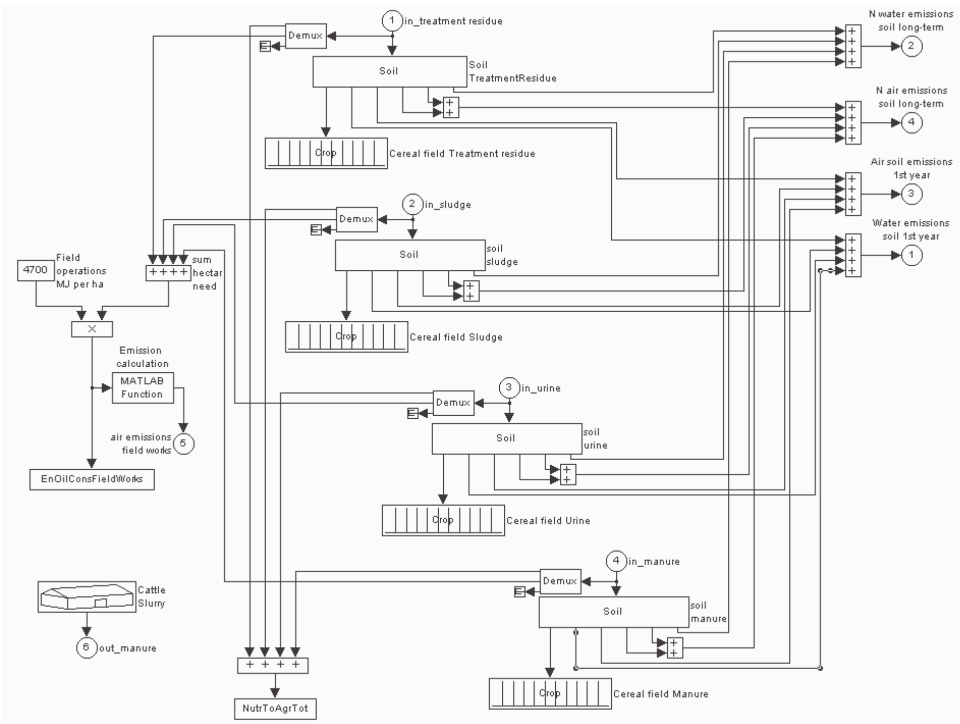

Figure 21. The agriculture sub-model.

The *soil* sub-model (Figure 22) calculate the fate of nitrogen in organic fertilisers spread on agricultural land. The nitrogen is divided into nitrate (NO_3^-), ammonium (NH_4^+), and organically bound nitrogen. Plant availability and emissions to air and water are determined. It is assumed that ammonium and nitrate are available to crops to the same degree as mineral fertilisers, except that some ammonium is volatilised as ammonia within the first hours after spreading. Organic nitrogen is only plant available after it has been mineralised to ammonium or nitrate. In the model, this mineralisation is divided into two time frames – the year of spreading and the remaining time. Thus, both immediate and long-term effects of the organic fertiliser can be evaluated. The model uses a budget concept, assuming that all nitrogen added is eventually degraded and become either plant nutrient or emission (Clarholm, 1997). It is assumed that 30 % of the organically bound nitrogen is mineralised in the first year, i.e. counted as plant available nutrient. Thirty percent of the organic nitrogen not mineralised in the first year is also counted as fertiliser; the rest is emissions in the form of N_2, N_2O or NO_3^-. This model is described in detail in Dalemo et al. (1998).

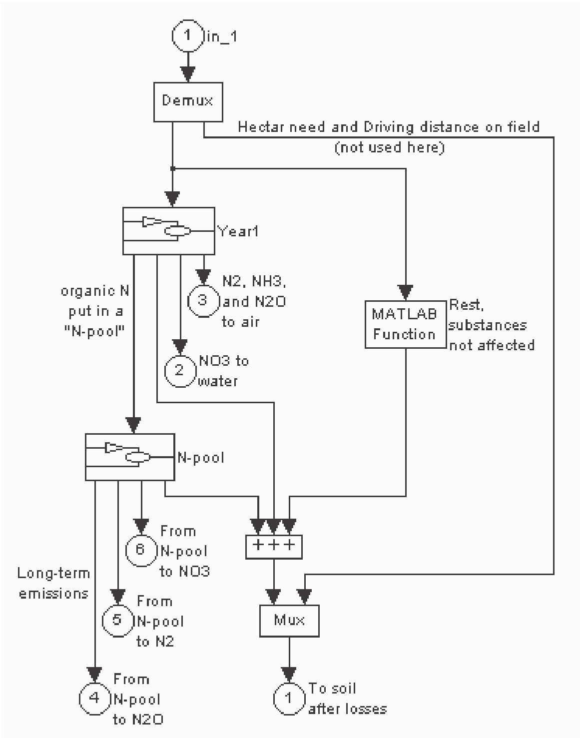

Figure 22. The soil nitrogen turnover sub-model.

The *grain production* sub-model calculates the expected yield of grain as a function of the amount of nitrogen applied, using a formula presented by Mattsson (1988). The formula is derived as an average nitrogen yield response from many field trials in Sweden. It was assumed that all other nutrients were supplied in adequate amounts. The sub-model, developed by Jönsson (1999), is shown in Figure 23.

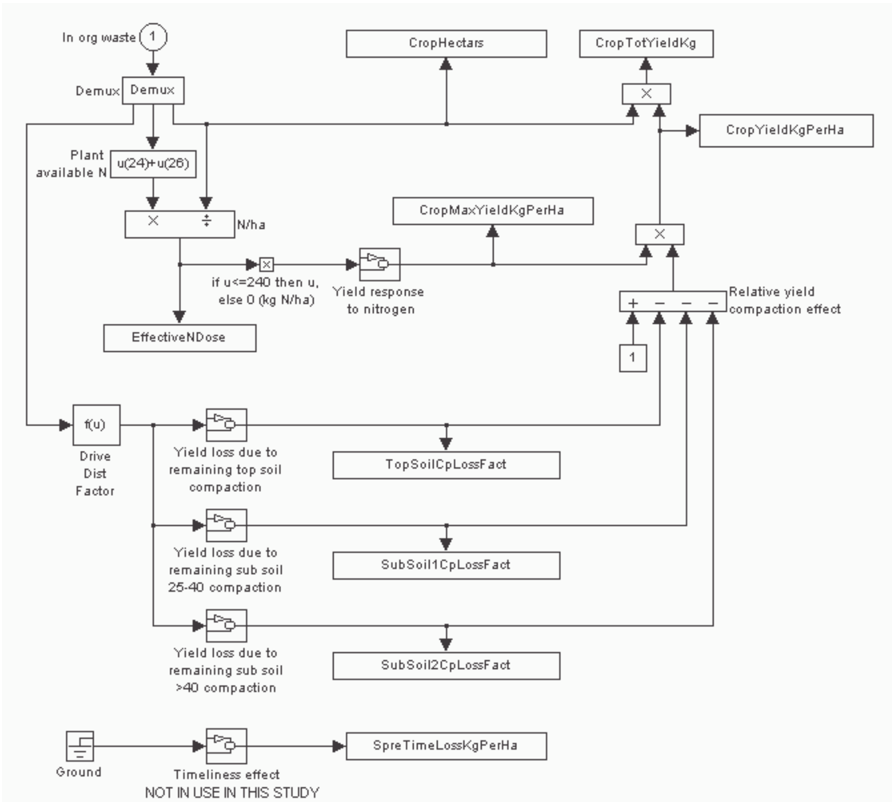

Figure 23. The grain production sub-model.

In-data are the ammonia and nitrate calculated as plant available in the preceding soil-model. From the maximum expected yield, the losses due to soil compaction and timeliness effect can be subtracted. In this study, only the soil-compaction losses were included. The soil compaction is calculated for the topsoil, the subsoil between 25 and 40 cm depth, and the subsoil below 40 cm depth respectively (Arvidsson & Håkansson, 1991, 1992). Variables are number of passes made by the implement, working width of the implement, water content of the soil, weight on the different axles and tire pressures of the different axles. For the topsoil, the clay content is also used as a variable. The compaction in subsoil is only increased by axle loads larger than 4 000 kg in the 25-40 cm layer and by axle loads larger than 6 000 kg below 40 cm. The losses due to timeliness effect were neglected in this study due to the large differences in material spread and in spreading time of year. Outputs from the sub-model are total yield of grain per year, and total yield per hectare and year. Other important aspects for potential yield of grain, such as phosphorus and organic matter, are not considered in this model.

Data for *field-operations* energy consumption are derived from Sonesson (1993). Some modifications were made about number of operations and power requirement. The modifications resulted in a somewhat lower energy consumption, 4700 MJ per ha, compared to the originally given 6240 MJ per ha.

Results and discussion of the organic waste management study

In the organic waste management study, I occasionally present results both for treatments of the household residues only (wastewater and household wastes from 32 households), as well as for all residues (household residues plus manure from 15 dairy cows and their young stock). In practice, it is probably not possible to treat only household residues in a digestion chamber or in a liquid compost reactor, due to the low dry matter content of these residues. However, to get a better understanding of the results and to show the household residues part in the entire system studied, the results are presented as if these treatments were possible.

Plant nutrients, dry matter, and water in waste fractions

The wastewater and household wastes from the 32 household together held about 17 % of the total input amount of nitrogen, while manure from the 15 dairy cows and their young stock contained the rest (Table 9). For phosphorus, the household share was about 13 % and for potassium, it was about 6 %. For practical and hygienic reasons, it was assumed that only the blackwater and household wastes were treated in the reactor treatments (together with the manure). Greywater contained more than 80 % of all water in the system and it was relatively unaffected by pathogens, so it could best be treated in an on-site sludge separation and infiltration construction or maybe used as irrigation water. This meant that about 15 % of the nitrogen and one third of the phosphorus in the household wastewater were “lost” with the greywater. However, the amount of phosphorus in greywater is highly dependent on which detergents are used for laundry and washing-up. In Sweden, most of the detergents used are of a low-phosphorus type. When the manure was included in the calculations the greywater held only a few percent of the total input of nitrogen and phosphorus.

Table 9. Plant nutrients, dry matter and water from different sources included in the study. *(percent of total input)* [1]

	greywater	blackwater	urine (incl. in blackwater)	household wastes	manure
N-tot	3	12	9	2	83
N-NH4	0	17	13	0	82
P	4	7	4	2	87
K	1	4	2	1	94
Dry matter	5	6	3	6	84
Water	88	5	1	0	7

[1] The sum is not always 100 percent due to rounding-off

Plant nutrients recycled to arable land

The different scenarios gave, as expected, large differences in performance concerning recycling of plant nutrients to arable land, when residues from households only were considered (Figure 24). Due to the relatively small contribution of household residues to the total, the difference in performance was less obvious when the manure was included in the simulations (not shown).

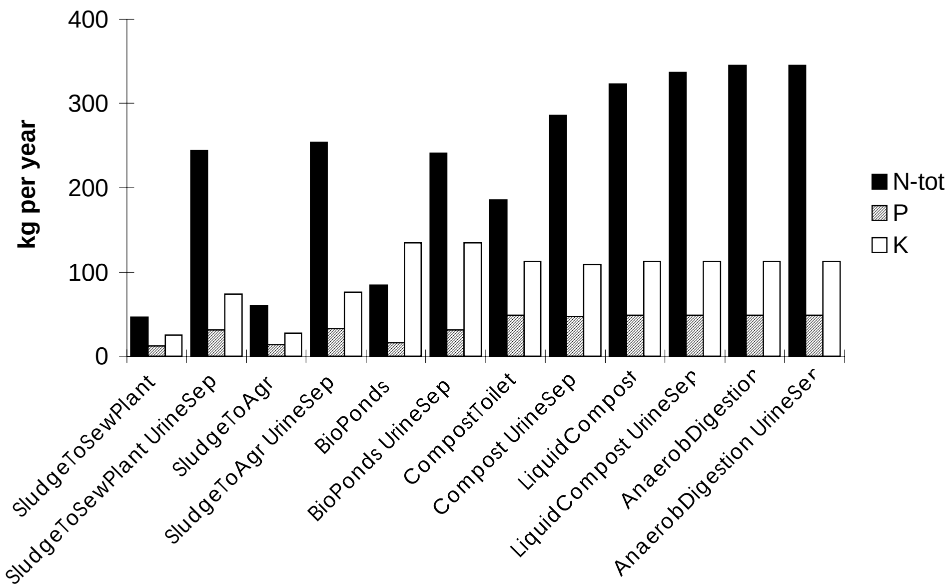

Figure 24. Plant nutrients recycled to arable land. Treated wastewater and household waste from 32 households. *(kg per year)*

The reactor treatments recycled much of the nutrients while the on-site sludge separating treatments gave large losses primarily of nitrogen but also of phosphorus and potassium (Figure 25). The biopond scenario gave large losses of nitrogen and phosphorus but the recycling of potassium was the largest of all scenarios. This owed partly to the fact that the greywater was included without sludge separation in the treatment, and partly to a probable under-estimation of potassium losses in the model calculations (no reduction was calculated). In the dry compost, it was mostly nitrogen which was lost. Condensation of outlet gases would recover most of the ammonia and thus give better results for this treatment. The difference between the liquid compost and anaerobic digestion in amount total nitrogen recycled was a result of post-treatment storage losses. During both treatments, almost no nitrogen was lost.

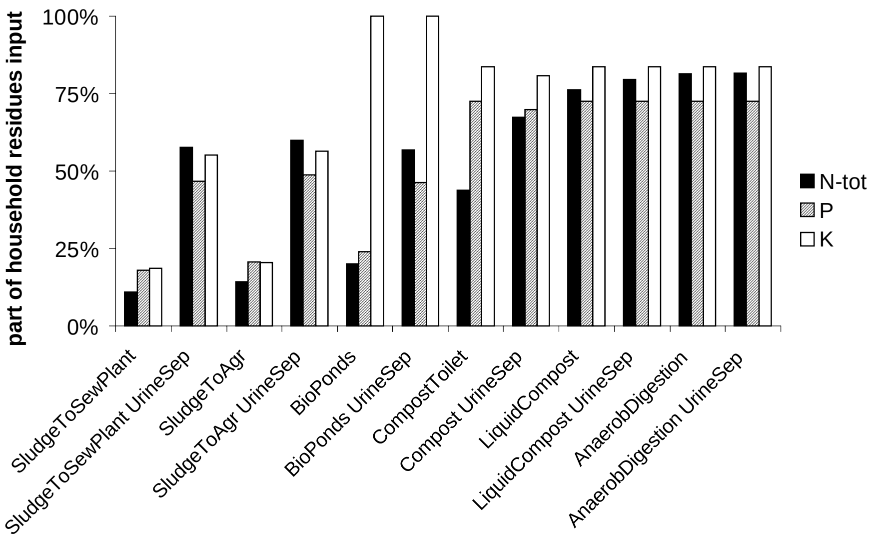

Figure 25. Plant nutrients recycled to agriculture, proportion of original content in household wastewater and waste. *(percent of original content in household residues)*

Urine separation; nutrient recycling

An introduction of urine-separating toilets gave a substantial increase in nutrient recycling in the on-site sludge separating and biopond scenarios. In the dry compost scenarios, urine separation resulted in increased nitrogen recycling rate, while the recycling of phosphorus was not affected. The amount of nitrogen delivered to arable land was increased in the two reactor treatment scenarios as well but to a minor degree since these treatments already retain the larger part of the nitrogen.

Additional yield of grain

The purpose of organic waste recycling is to utilise the nutrient and organic matter resources of the waste, i.e. the recycled residue is used as organic fertiliser in order to give larger yields of crops and possibly to upgrade the soil fertility. Thus, for the farmer, the total amount of recycled plant nutrients is not that relevant, since different fractions of nitrogen in particular have very different fertilising properties. Instead, the expected increased yield is important. Furthermore, different waste treatment methods produce residues of varied contents (Figure 26).

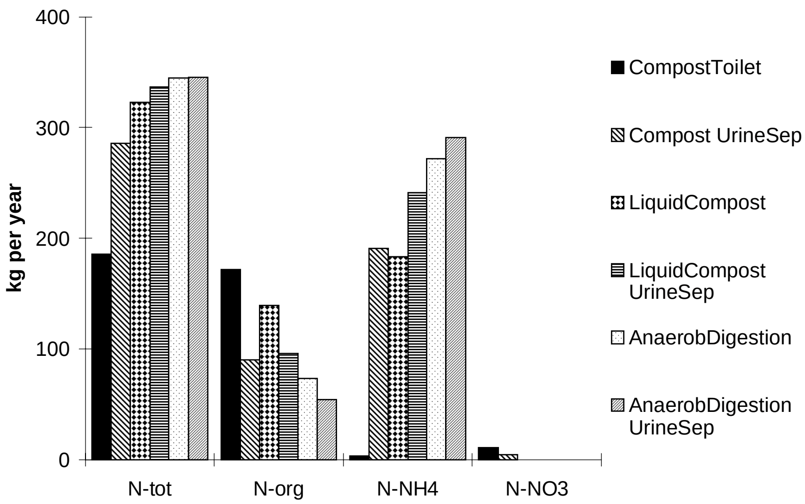

Figure 26. N-fractions recycled to arable land in scenarios with blackwater separating systems. Treated wastewater and household waste from 32 households. *(kg per year)*

To be able to assess the fertilising quality of the organic wastes, a model that calculated nitrogen plant availability and nitrogen losses in soil was included. In addition, to make the results more comprehensible a model that calculated the expected additional yield of grain due to the recycled plant-available nitrogen was also included. The model used for the additional yield calculations was quite crude since it did not include phosphorus, potassium, organic matter, or water. It rather assumed that P and K were supplied in sufficient amounts. Despite the model's imperfections, the results were reasonably appropriate for short-term yield estimation since the rate of recycling of both P and K point in the same direction as for N, i.e. the growing demand for P and K when more N was supplied, was probably met. For most soils, phosphorus, potassium and organic matter are more a matter of long-term soil fertility and are not easily connected to the next year's yield. However, for long-term yield estimation (which is the really interesting knowledge) these should be included.

The calculations of soil losses, plant availability and additional yield of grain gave some changes in results compared to the amount of plant nutrients recycled (Figure 27).

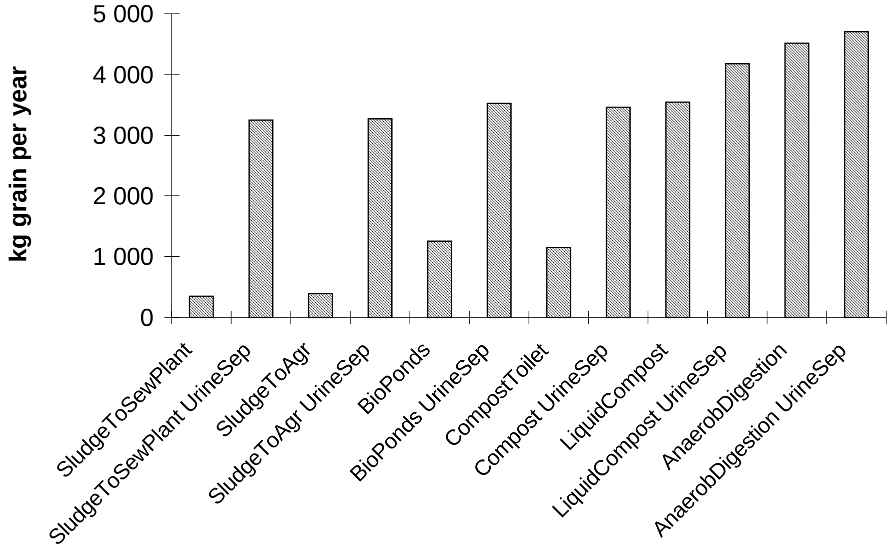

Figure 27. Expected additional yield of grain as a result of recycling residues from 32 households to arable land. *(kg per year)*

Dry compost residues, for example, have a high content of organic nitrogen, which was assumed to have about 30 % fertilising efficiency (30 % mineralised first year, plus 30 % in the long-term of the small fraction that remains after emissions in organic N).

It is noteworthy that an introduction of urine-separating toilets in the on-site sludge separation, biopond and compost scenarios gave a significant rise in the expected yield of grain, up to the level of the liquid compost scenario.

Concerning yield of grain in the biopond scenarios, the figures could be an overestimation, since the losses of nitrogen when spreading the water in this case were not accounted for (and nitrogen was the only substance included in the calculation). On the other hand, the rise in yield due to the large amount of water irrigated was not accounted for either. In practice, the water is spread on growing crops, often at night when the evaporation rate is low. Altogether, it is likely that the model underestimated the expected yield of grain, but this has to be further investigated.

The differences in expected yield of grain between scenarios were large when looking at the household part only but the results were much more levelled out when the manure was taken into account (Figure 28).

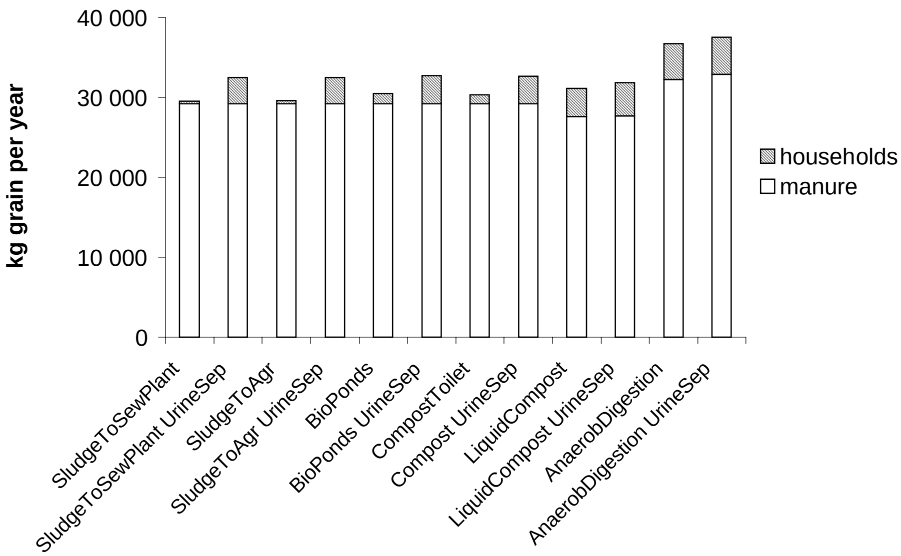

Figure 28. Expected additional yield of grain due to the recycling of organic waste to arable land. Treated wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock. *(kg per year)*

The difference in expected grain yield between liquid compost and anaerobic digestion residues is somewhat surprising. It is partly an effect of calculated differences of losses during post treatment storage. The liquid compost residue lost 12 % of the ammonia, while the anaerobic digestion residue lost only 1 %. Of more importance is probably the fact that the aerobic compost process built up more organic material in micro-organisms than the anaerobic process and thus had more nitrogen bound up in the organic matter. However, all organically bound nitrogen was treated equally in the model (30 % mineralisation in the first year). This was probably a good estimation for the dry compost residues that are well degraded, but for the liquid compost residue, the yield might have be under-estimated, since this organic nitrogen should be more easily degradable than that in dry compost residues. The difference between digestion and liquid composting could also be an effect of the longer retention time in the digestion, resulting in more breakdown of organic material, which gave a higher content of ammonia. More knowledge about the performance of liquid compost reactors and the utilisation of liquid compost slurry is needed for the improvement of the model.

Heavy metal recycling rate and contamination of arable land

As can be seen in Figure 29, the manure contained a large part of the heavy metals present in the system. Exceptions were mercury (Hg) and chromium (Cr). A possible recycling of organic waste from households should thus contribute a fairly small amount to heavy metal contamination. Though it looks small, I will discuss its importance in the following. It should also be kept in mind that the heavy metal content in both manure and household residues may vary a lot, so the household residues part may be larger in some cases.

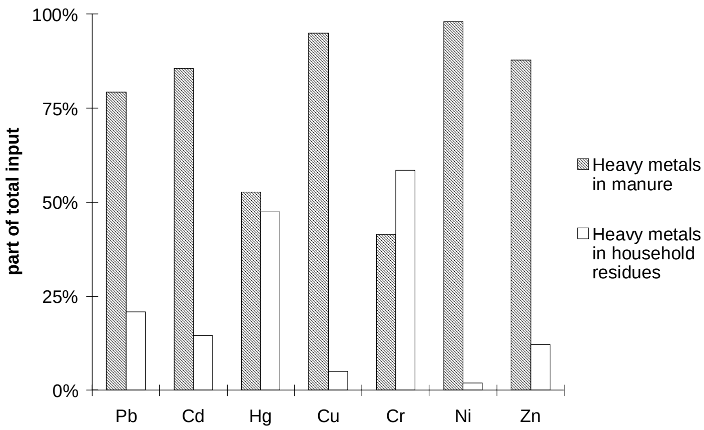

Figure 29. The distribution of heavy metal content in manure from 15 dairy cows and their young stock, and wastewater and household waste from 32 households. *(part of total content in sources)*

Recycling of heavy metals from household residues

All scenarios having blackwater-separating systems recycled more plant nutrients to arable land. The logical negative side of this would be that more heavy metals would be fed back to soil as well. However, the results indicated that this was not at all that evident (Figure 30). The biopond results are not displayed due to lack of relevant data. Presumably some of the heavy metals are caught in the sediment at the bottom of the bioponds, but to which extent this occurs is impossible to assess from the existing data. However, there is a risk that this system recycle more heavy metals to arable land since the greywater (after treatment) is recycled together with the rest of the sewage water, which it is not in the other systems. More research is needed to be able to evaluate the performance of the biopond system concerning heavy metal contamination of arable land.

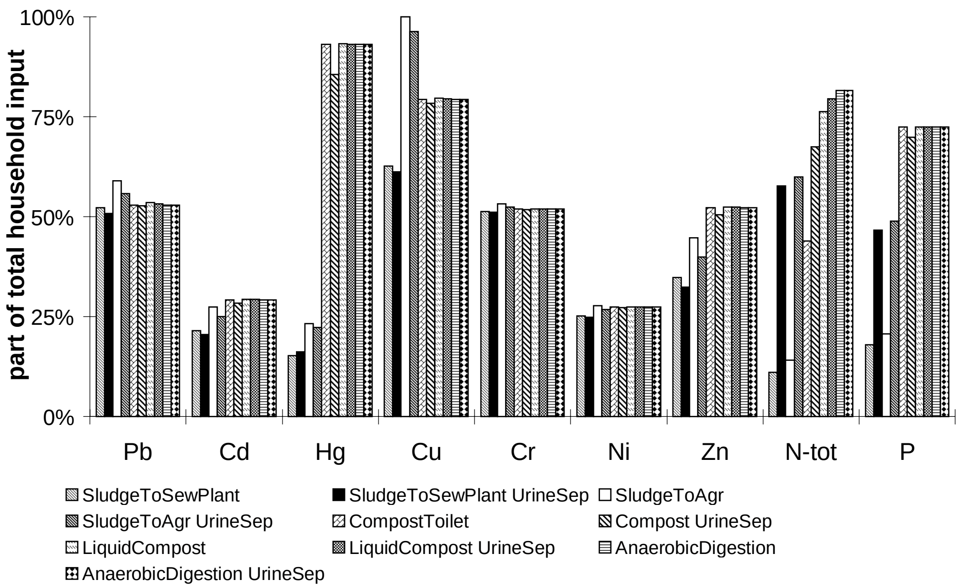

Figure 30. Heavy metals and plant nutrients in household wastes and wastewater recycled to arable land. *(part of initial content in household wastes and wastewater)*

It can be noted that the different metals were recycled to different extents. In this study, it was not possible to establish whether that was the result of unreliable data, or whether it was a true difference (with exception of the Cu, which is discussed below). However, most of the results can be explained by studying the initial content of heavy metals in the different waste fractions (Table 10). The rate of recycling for all metals except copper followed more or less the initial content in faeces and household wastes. The major uncertainties concerning the heavy metals originated from the sources themselves (content in household waste and wastewater) and from the sludge separation sub-model. Note the low content of heavy metals in urine. This, together with its relatively high content of plant nutrients and low content of water (see Table 9**),** explains the attention urine separation has received in Sweden in recent years.

Table 10. Heavy metal content in different fractions of household wastewater and waste. *(percent of total)* [1]

	greywater	urine	faeces	household waste
Lead (Pb)	53	0.1	1	46
Cadmium (Cd)	76	0.5	8	16
Mercury (Hg)	14	2.9	75	8
Copper (Cu)	63	0.8	11	26
Chromium (Cr)	50	0.2	1	49
Nickel (Ni)	75	0.2	2	23
Zinc (Zn)	57	0.1	18	25

[1] The sum is not always 100 percent due to rounding-off

Concerning the (too) large figures for the copper (Cu) recycling ratio in SludgeToAgr-scenarios, they were probably the result of the data used emanating from different sources. Wastewater data were Swedish averages, while data used in the sludge separation sub-model were measured contents in sludge from places with limy soils that give hard water (measured in °dH). This kind of water often gives a high content of Cu in sludge, as the carbonate ions that accompany the calcium release copper from hot-water pipes used in houses. Thus the model probably over-estimated the content of copper in the sludge. Some regions in Sweden with hard water have problems with high copper contents in sewage sludge. Several solutions may be at hand. Central installations for decalcification of the water have been discussed and investigated in e.g. Uppsala. The problem here is to remove the calcium and the carbonate without also removing magnesium (Mg) and other microelements good for human health. Calcium is good for health but it is judged that it is of greater importance to get rid of it due to the copper problem. Furthermore, it increases the consumption of soap and detergents for washing etc. and it may cause calcium-coating problems in water pipes and different equipment. Ordinary decalcification-installations that use salt to substitute the calcium ion for a sodium ion do not solve the problem of copper release to water, since the carbonate-ion not is removed. This type of equipment also removes the magnesium.

Another option is to use other materials in water pipes. This is too expensive to be done in the short term and no guarantee can be given that the new materials will not give rise to new problems. A third possibility is to use blackwater-separating systems. This is also expensive, but would probably solve the problem with Cu and other contaminants, since the copper released from hot-water pipes would not affect the matter destined for use as an organic fertiliser. Additionally, different detergents and chemicals ending up in wastewater largely originate from greywater. The greywater would then have to be treated in separate systems.

In this study, it was assumed that sludge from greywater was recycled in the blackwater separating scenarios. The data were unreliable but results (not shown) indicated that it was a question of small amounts. The exception here was also copper, probably for the same reason as mentioned above. An assumption that Cu in greywater-sludge was at levels equivalent to the other metals meant that the rate of Cu recycling followed the content in household wastes and faeces.

Similar errors, such as those discussed for copper above, may be valid for the other metals. For example,. it might be behind the result for mercury (Hg), which is found in faeces and probably originates from dental amalgam. According to Figure 30, much Hg was released with the outlet water from sludge separators (a small amount was caught in the sludge). This could be the case, but it depends on the form in which the mercury is present (metallic or organic). In organic form, it is hard to get into solution. In metallic form, it might be more loosely bound or adhered to particles and/or organic matter (Wikberg, 1999). Results presented in Kärrman et al. (1999) indicate that some 60 % of Hg is caught in the sludge from a sewage plant using chemical precipitation. Since no chemicals are used in the sludge separators, less mercury could be trapped in this case. However, knowledge in this area seems to be sparse.

Heavy metal contamination of arable land

A very important question is to what extent the recycling of residues involves a heavy metal contamination of the arable land. This is very hard to assess without knowledge about type of soil, type of crop, and other site-specific variables. An example of such a balance calculation is shown in Appendix 9. It shows the amounts of heavy metals both recycled to arable land and the possible export with the grain produced on that land. The grain included is the total grain production, i.e. the potential additional grain-production from the organic waste recycling plus the base production produced with no fertilising. The difference between import and export is reported as net soil contamination. However, it should be remembered that the contribution from precipitation and other possible sources was not taken into account. For example, in cadmium contamination of soils in Sweden, precipitation is a large contributor (Hedlund et al., 1997).

Energy turnover

An important aspect in the choice of treatment system is the energy turnover. I hoped to find systems that not only had a low energy demand, but were also energy self-sufficient or even net energy producers. None of the scenarios studied managed to fulfil this goal (Figure 31) when the energy consumption for grain production field operations was included. The anaerobic digestion treatment with a urine-separating toilet system and no hygienisation of incoming material was closest to reaching the energy self-supporting goal, but needed to decrease its energy consumption by a third. It must be remembered when studying the diagrams that energy is presented as if all energy had the same quality, which is not the case. For example, the liquid compost produced heat ("low-quality" energy) and consumed electricity and fossil oil ("high-quality" energy). Furthermore, the waste-transport distance chosen in this study was fairly short (2 km). A sensitivity analysis where the distance was set to 10 km was also performed. It is presented last in this chapter.

The (fossil) oil consumption arose in transportation and spreading of the material, and in the grain-production field operations. The heat and electricity net turnovers originated from energy consumption and production in the different treatments. In addition, the production of drinking water (used in households for washing, flushing of toilets etc.) consumed both oil for transport and electricity for pumping and production of chemicals. In the biopond scenarios, electricity was also consumed by the pump-and-pipe transport of both incoming wastewater and treated water delivered to irrigation of arable land.

The anaerobic digestion was run in two modes (with or without hygienisation of incoming material) to show the potential energy production. Both choices are possible in practice. It is a question of the hygienic standard of incoming material and of where the residues are going to be spread. Another hygienisation option might be extended storage of the treated residues. In Sweden, 6 months storage is generally accepted as a hygienisation method.

Also included in the energy consumption diagrams was the energy needed for the production of extra (fill-up) nitrogen fertiliser to get the scenarios to produce equal amounts of grain. Where the energy turnover for handling of household residues only was presented (Figure 32), energy consumption for an extra area of grain production also had to be included in all but two scenarios. This extra area was necessary since addition of nitrogen up to the desired maximum yield on the acreage fertilised with treated organic wastes was not enough to produce sufficient yields of grain. The extra area was conventionally managed, i.e. mineral fertilisers were used.

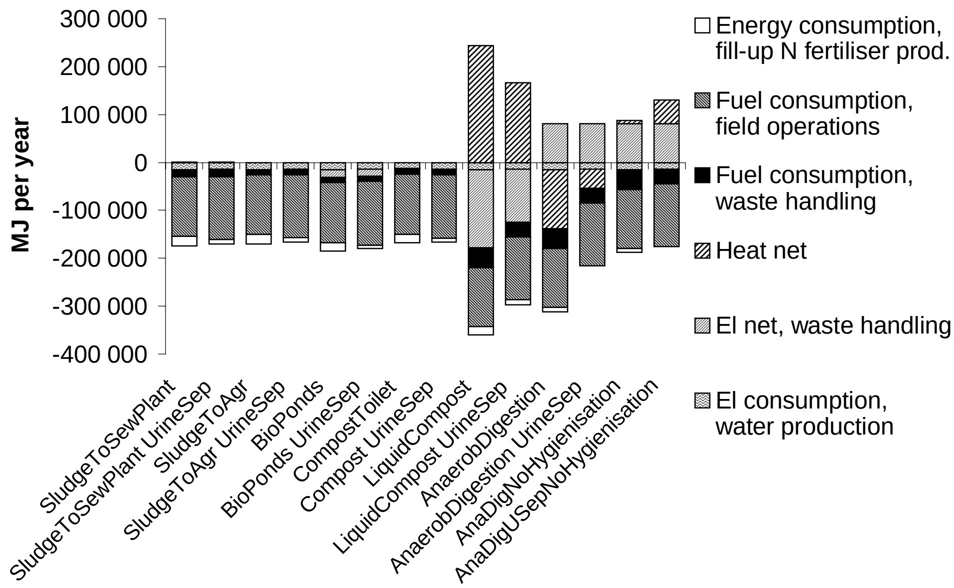

Figure 31. Energy turnover for treatment of wastewater and household waste from 32 households and manure from 15 dairy cows and their young stock, including water-production and grain-production field operations. *(MJ per year)*

The compost treatments produced heat that was not easily utilised. It demanded insulated reactors and heat-exchange systems. In the dry compost, I assumed the heat to be ventilated away, but in a reactor-compost, it could be captured by condensation of the outlet gases. Concerning the liquid compost, it is probably not possible to take any heat out from the reactor since the heat produced is needed to heat incoming slurry. The main part of the heat can instead be drawn directly after the slurry is taken out of the reactor. There are no readily available solutions for doing this (Skjelhaugen, 1997) but I still assumed that the heat produced could be utilised. Thus, since the data and the assumptions made were somewhat uncertain, the possible utilisation potential of the heat might be overestimated in these scenarios.

Concerning the energy turnover results, both liquid compost scenarios gave small net energy surpluses (Figure 31) if agriculture was not taken into account. When only the household residues were taken into account (Figure 32), only the liquid compost scenarios showed results close to break-even energy turnover. All other scenarios showed clearly negative energy turnovers.

Effects of urine separation are discussed below in sub-chapter “Urine separation; energy consumption and transport”.

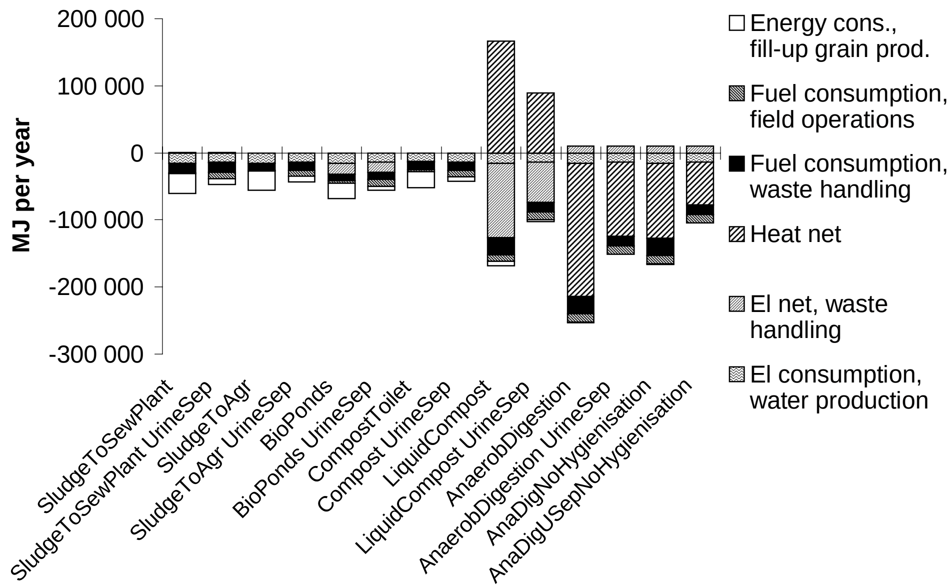

Figure 32. Energy turnover for treatment of wastewater and household waste from 32 households, including water-production and grain-production field operations. *(MJ per year)*

Biogas from anaerobic digestion

The biogas produced by anaerobic digestion can be utilised in at least three different ways. Figure 31 and Figure 32 present the case where the gas was used as fuel in a stationary combustion engine that produced electricity and heat. The biogas can also be cleaned and compressed for use as vehicle fuel and it can be burnt for heating of e.g. buildings.

Figure 33 shows the energy value of the methane produced and the actual energy consumption when manure is included in the system. If the gas is purified and compressed for use as vehicle fuel, the loss is about 7 % of the initial energy content (Dalemo, 1998). The remaining gas would in that case hold about 295 000 MJ annually, corresponding to some 8.3 m^3 diesel.

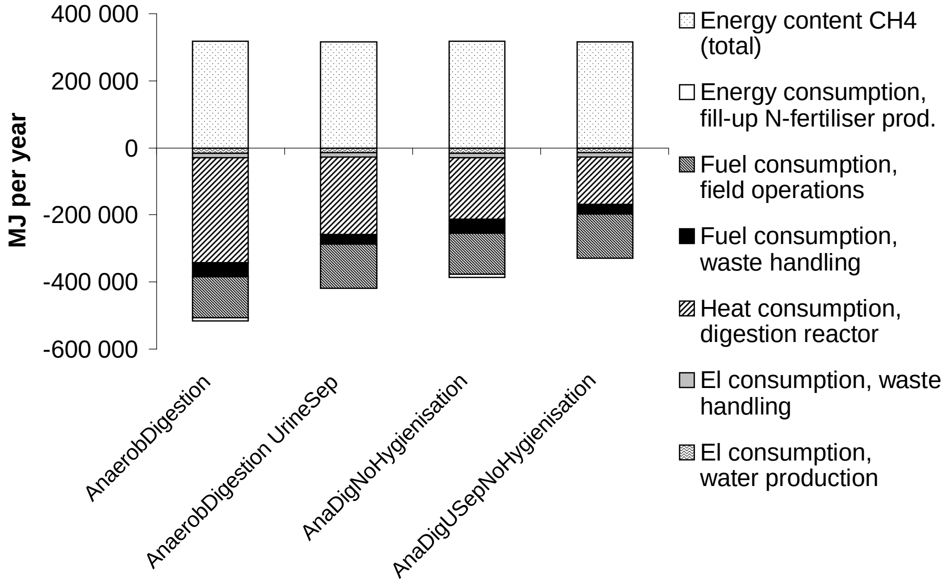

Figure 33. Energy turnover for anaerobic digestion treatments of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock. Energy value of methane produced compared to actual energy consumption. *(MJ per year)*

All anaerobic digestion scenarios had energy deficiencies, but the scenario with urine-separating toilets and no hygienisation of input material showed a result that at least came close to break-even. The energy self-supporting point would be reached if the energy consumption in grain-production field operations were decreased by approximately 10 %, which is possible if for example the agricultural soils were lighter (sandy). Another possible option is to decrease both fuel consumption in waste management and clean-water production energy consumption, by respectively a sixth and two thirds.

A theoretical option is to use some of the gas in a stationary engine, producing electricity and heat needed for internal use, and to purify and compress the rest for use as vehicle fuel. About 75 % of the gas would have to be used in the stationary engine to produce enough heat. That would give an electricity net production of about 40 000 MJ/year (some 11 000 kWh). The gas left for vehicle fuel would supply about 60 % of the field operation's energy requirement or about 45 % of the total energy requirement for transport and field operations in the system.

The economics of a small-scale digestion system with equipment for gas purification and compression were not studied here. The high investment cost would probably make this alternative possible only for larger settlements, or the investment may have to wait until the fuel price has risen substantially.

Urine separation; energy consumption and transport

It has often been said that the introduction of urine-separating toilets would result in more transportation work. According to this study, this is not the case, which is shown in Figure 31, Figure 32, Figure 33 and Figure 34. Most treatments showed about equal energy consumption for systems both with and without urine separation. In anaerobic digestion scenarios, urine separation resulted in lower energy consumption for heating the digestion reactor as well as for transportation. The same was valid for the liquid compost treatment. The largest energy saving effect was obtained in these treatments internally. The smaller amount of water treated resulted in lower electricity consumption (approx. 30 %) in the liquid compost reactor and lower heat consumption (approx. 25 %) in the anaerobic digestion chamber. Concerning the household residues only, the fuel consumption for transportation and spreading was decreased by almost the half due to the water saving achieved in the urine-separating toilets in liquid compost and anaerobic digestion treatment scenarios (data not shown).

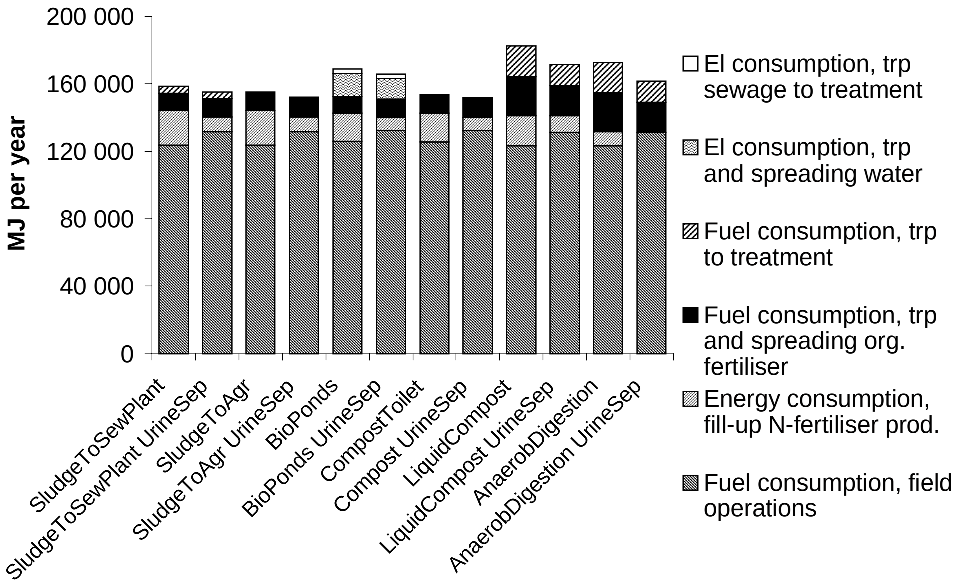

Figure 34. Energy consumption due to transportation and spreading of wastewater and household waste from 32 households and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(MJ per year)*

Figure 34 shows that the energy consumption for transporting the organic wastes to arable land and spreading it was somewhat larger when urine separation was used in systems that did not have vehicle transportation of sewage water. That was, however, compensated for by lower energy consumption in other parts of the systems.

Energy efficiency

One major question is, which system is the most efficient in terms of energy use. A logical measure could be energy turnover (production minus consumption) per kg additional yield of grain produced (Figure 35). However, since the energy turnover figures did not take the energy quality into account, one has to be cautious when interpreting the results. For example, the height of the liquid compost columns (Figure 35) was not of same value as that of the anaerobic digestion columns. The former comprised heat production and electricity consumption values but the latter contained heat consumption and electricity production values.

Another possible measure could be MJ energy turnover per MJ in grain produced (or its inverse). That has, however, not been calculated, but could easily be done by multiplying kg grain (presented in Figure 35) by its energy value, which is 18-23 MJ/kg (Fluck, 1992).

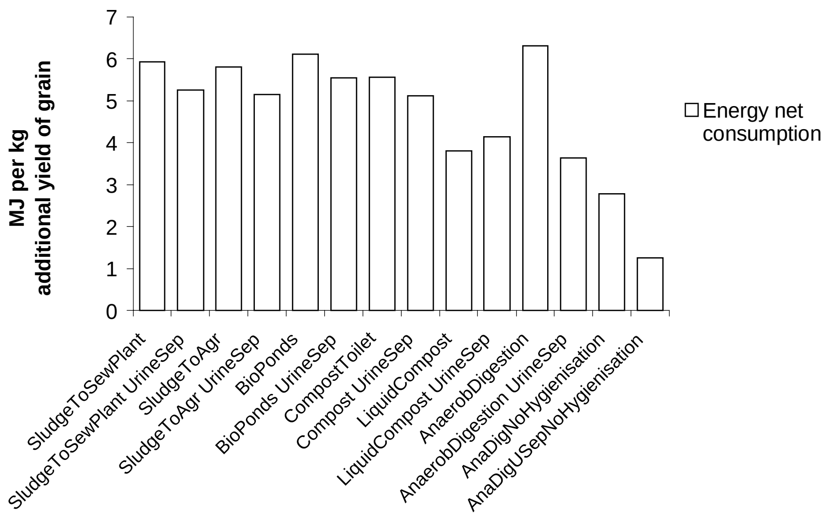

Figure 35. Energy net consumption per additional yield of grain. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(MJ per kg grain)*

Studying the four anaerobic digestion scenarios gives rise to some questions. Anaerobic digestion seemed to be capable of being a very efficient treatment method for the residues in the system studied, but it also showed one very negative result. One possible interpretation of this could be that it is important not to choose an “efficient” treatment only, but also to consider other parts of the system in order to make the whole system work efficiently. Another interpretation could be that, to be efficient, anaerobic digestion should be combined with urine-separating toilet systems and, if possible, the incoming wastes should not be hygienised. The liquid compost results were, as mentioned before, not of very great reliability due to imperfections in the model concerning heat production.

Primary energy resources consumption

When resources were the focus, the consumption of primary energy resources (Figure 36) was a better measure than energy consumption since it tried to assess the amount of resources consumed under the whole lifecycle of the energy carriers. Here, the result is presented as MJ primary energy resources per kg additional yield of grain.

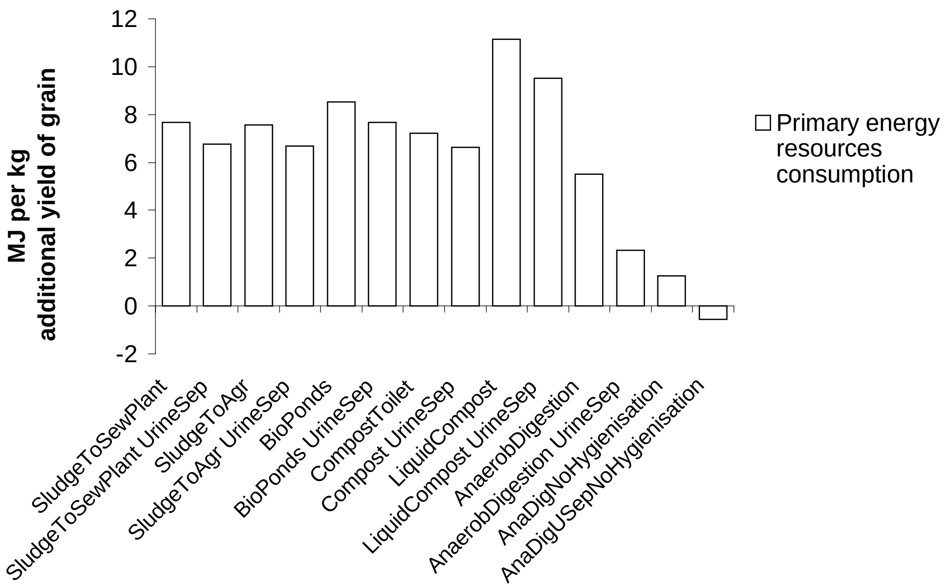

Figure 36. Primary energy resources consumption per additional yield of grain. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(MJ per kg grain)*

When calculating consumption of primary energy sources, biopond and liquid compost scenarios that used much electricity stood out a bit more since electricity uses more primary energy resources than fuels do. For the same reason, the anaerobic digestion scenarios showed better results here. The negative column for the last scenario (Figure 36) denotes that this scenario actually produced a primary energy surplus. This was possible (although the energy turnover was negative) since the gas produced was used for production of electricity (and heat), of which the treatment consumed some but the large part was delivered to the electricity grid. The electricity assumed to be replaced was Swedish average electricity, which consumes 2.35 MJ primary energy resources per MJ electricity delivered.

Eutrophication

Six substances were included in the assessment of eutrophication. These were four water-emissions (ammonium (NH_4^+), nitrate (NO_3^-), phosphorus (P), and organic matter measured as chemical oxygen demand (COD)) and two air-emissions (ammonia (NH_3) and nitrous oxides (NO_x)). A maximum contribution scenario was used, i.e. both phosphorus and nitrogen contributed to the eutrophicating impact. However, in most aquatic systems one or other of them is limiting.

The eutrophicating impacts are shown in Figure 37. Emission of nitrate from soil was the largest contributor to eutrophication in all scenarios. Slightly more than 80 % was leached during the first winter after spreading, the rest was a long-term impact. It is interesting that the results for liquid compost versus anaerobic digestion were reversed when the soil emissions were taken into account. This might have been a true difference but it could also have been an effect of imperfections in the model. Nitrate that is lost by leaching emanates from organically bound nitrogen. Since the model treated organically bound nitrogen in all different substrates equally, it might have over-estimated the nitrate emissions from liquid compost residues.

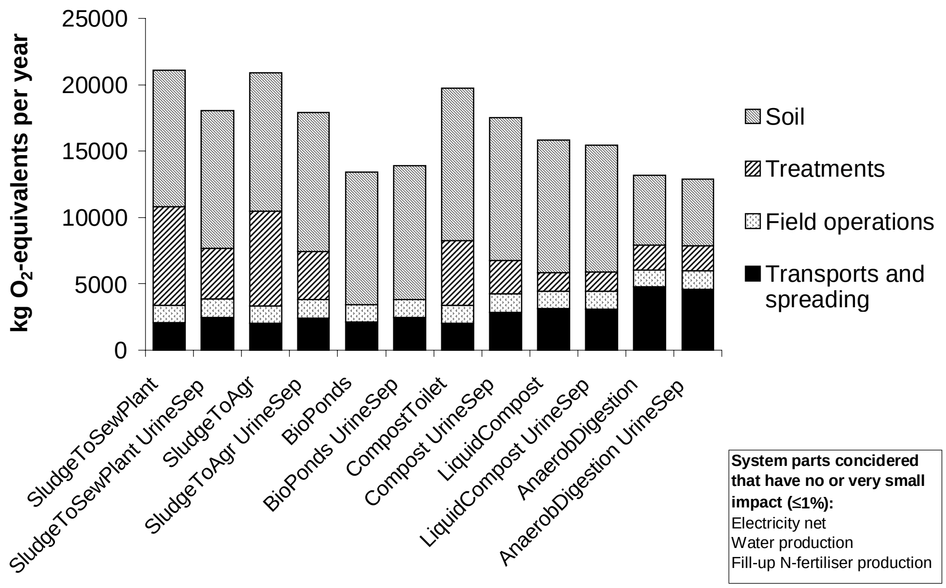

Figure 37. Eutrophication, maximum scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(kg O_2-equivalents per year)*

Concerning the impact from transport and spreading, ammonia emissions in connection to spreading of the residues were the dominating contributor. Since anaerobic digestion residues contained more ammonium than e.g. liquid compost residues, more was also lost as ammonia.

For treatments, phosphorus (P) and nitrate (NO_3) emissions together were dominant in the four scenarios having on-site sludge separation of sewage water. An introduction of urine-separating toilets here reduced P-emissions by about 35 % and NO_3 emissions by 70 %. In the scenarios which had blackwater-separating sewage systems, where only greywater was released to recipient waters via sludge separation and infiltration, the situation was somewhat different. Here, phosphorus dominated the emissions from treatments. Nitrate had about one third of the impact that phosphorus had. The higher bars for the dry composting scenarios concerning treatment (Figure 37) were the result of ammonia emission from the compost. In the dry compost scenario without urine separation, NH_3 actually had a three times larger eutrophicating impact than P – when urine-separating toilets were installed, these impacts were about equal. The biopond treatment resulted in zero emissions to water (from the treatment) since all water was treated and used for irrigation. In practice, there may be leakage from pipes and storing reservoirs but that was not considered in the model.

Global warming

Three greenhouse-gases were included in the global warming impact assessment. These were carbon dioxide (CO_2), methane (CH_4), and dinitrogen oxide (N_2O).

The fossil CO_2 was emitted in the combustion of fossil fuel used for transportation work and for production of electricity. The CO_2 emissions caused by transportation of household residues were, as for the energy demand shown above, small in comparison to the emissions occurring due to spreading of the manure except in the liquid compost and anaerobic digestion treatment scenarios. For a system with longer transportation distances the emissions would of course be larger. But, following the discussion above about the energy self-sufficient biogas scenarios, digestion with no hygienisation of incoming material would emit almost no fossil CO_2 since all fuel needed would be produced internally from the biological wastes.

The CH_4 was produced in some of the different treatments and during storage. Data were lacking in some parts but the supposedly most important contributors were included in the models or will be discussed later.

The sludge separator sub-model did not produce any CH_4 emission, which the real construction surely does. Such production is, however, probably small if the sludge separator is emptied at least once a year. The liquid compost sub-model did not calculate any emissions either. In that case it was not the treatment in itself that produced the methane, but the post treatment storage. During storage, anaerobic conditions will occur, but it should take some time for the anaerobic (methane-producing) microorganisms to reproduce since the slurry is completely dominated by the aerobic microorganisms that have developed during the aerobic treatment. The amount of CH_4 produced should at least not exceed that produced in digestion residue storage (where the gas is collected and added to the gas produced in the reactor). The bioponds normally have aerobic conditions, and thus no production of CH_4. However, during hard winters, when the ponds and reservoirs get ice-covered, anaerobic conditions may occur. No data were available and it was very hard to even make an assumption of how much methane could be emitted under such circumstances. There are surely also anaerobic conditions in the bottom sediment during most periods of the year, but there have been no observations of gas bubbling up. When measured, no detectable concentrations of CH_4 were found in the air just above the water surface (Duveborg, 1997).

N_2O is mainly produced in soil processes, and to some extent in dry composting and production of nitrogen fertilisers.

The potential global warming impact is shown in Figure 38. The dominant part in the system was the grain production process, especially the field operations. CO_2 emanating from the combustion of fuels was a main contributor to global warming. Emissions N_2O from soil had a global warming impact but the model was quite crude in this respect, so the results have to be interpreted with caution. More research is needed. The impact from treatments was small except in the dry compost scenarios. In these, emissions of mainly N_2O and to some extent CH_4 from the compost caused the greenhouse effect. Electricity consumption made a minor contribution to global warming when the electricity was produced as Swedish average electricity. The small negative columns for the anaerobic digestion scenarios (Figure 38) reflect the emissions avoided due to the net production of electricity. The net result is thus the positive column subtracted by the absolute value of the negative column.

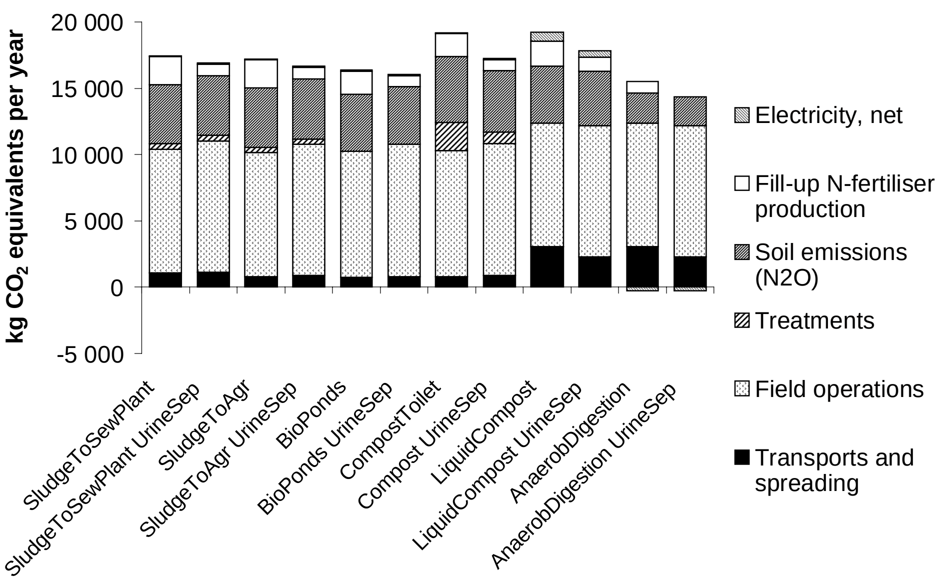

Figure 38. Potential global warming, 100 years scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain- production field operations plus additional production of nitrogen fertiliser. *(kg CO_2-equivalents per year)*

Acidification

Four substances were included in the acidification assessment. These were ammonia (NH_3), nitrous oxides (NO_x), sulphuric oxides (SO_x), and hydrochloric acid (HCl). Figure 39 shows the maximum acidification impact. Dominating contributors were NH_3 emissions in connection with spreading of treated organic waste and manure, and NO_x emission from fuel consumption in field operations.

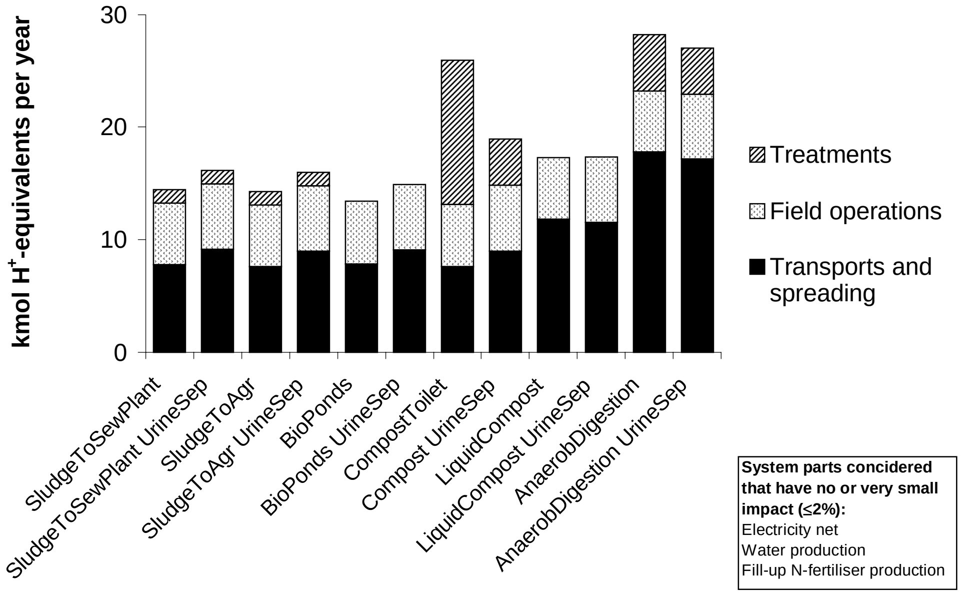

Figure 39. Acidification, maximum scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(kmol H^+-equivalents per year)*

However, in many ecosystems in Scandinavia and Northern Europe, nitrogen compounds are usually neutralised to a large extent. Therefore, the minimum acidification impact, where nitrogen compounds had no effect, is presented in Figure 40. Please note the difference in scale compared with Figure 39. In practice, the acidification impact is probably somewhere in between these results. As can be seen, the impact from treatments in anaerobic digestion scenarios was still relatively large since it was mainly an effect of emission of SO_2. The digestion plant in itself did not emit sulphuric oxides, but when gas was combusted, SO_x was emitted. The large acidifying impact from composting treatments stemmed from emission of NH_3, which had no impact in Figure 40.

Urine-separating systems gave larger ammonia losses in connection with spreading, which resulted in a slightly increased acidifying impact. Likewise, anaerobic digestion scenarios had more emissions of ammonia when digestion residues were spread, since some organic nitrogen in the manure when digested was mineralised to ammonium, which is more easily vaporised as ammonia.

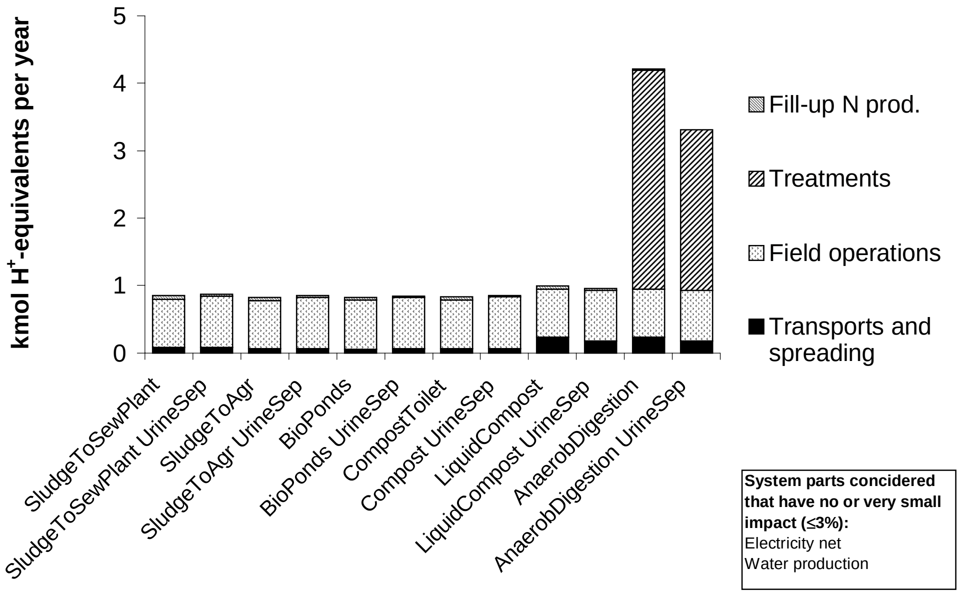

Figure 40. Acidification, minimum scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(kmol H^+-equivalents per year)*

Photo-oxidant formation

In the assessment of photo-oxidant formation, four gases were included. Three of these were categorised in photochemical ozone creation potentials (POCPs). These were methane (CH_4), non-methane hydrocarbons (NMHC), and carbon monoxide (CO). However, ozone creation can be limited by organic substances like those mentioned and/or by nitrous oxides (NO_x) (Lindfors et al., 1995). For large parts of Europe, it is actually expected that emissions of NO_x are the most important factor for ozone production (Axelsson et al., 1995). There is no categorisation factor developed for NO_x so this substance is presented in a separate diagram (Figure 42). The results for POCP are presented in Figure 41.

For POCPs, it was NMHC emissions from the combustion of fuel in grain-production field operations which dominated. However, the data used were valid for the combustion of diesel in lorries and the emission from tractor engines working under different conditions may vary a lot. According to Hansson et al.'s (1998) preliminary study, the emission of hydrocarbons (HC) varies between 0.041 g/MJ (for harrowing) and 0.238 g/MJ (for loading), but most of the values are below or just above 0.100 g/MJ. The model used 0.066 g/MJ, which should be reasonably appropriate, or it may be a slight under-estimation of the NMHC-emissions.

The larger impact from transport and spreading in the liquid compost and anaerobic digestion scenarios was due to the increased amounts of material (much water) that had to be transported and spread. The extra emissions of NMHC from combustion of fuels mainly caused the increase.

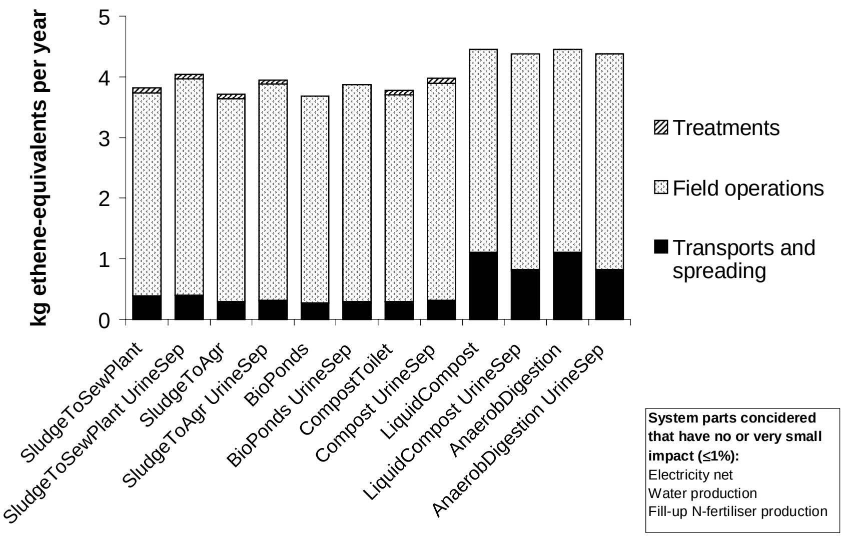

Figure 41. Photo-oxidant formation, using the concept of photochemical ozone creation potentials (POCPs). Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(kg ethene-equivalents per year)*

Concerning the emissions of NO_x (Figure 42), these were dominated by the emissions from field operations. However, the larger impact for the liquid compost and anaerobic digestion scenarios was due to the increased need for transportation and (in the case of anaerobic digestion) the emission of NO_x when biogas was combusted in a stationary engine for electricity production. The transportation in these systems could possibly be decreased by proper localisation of treatment plants, so that pumping of the slurry could be used instead of vehicle transportation. The emission of NO_x from combustion of the biogas was probably more difficult to avoid. Thus, in regions where nitrogen compounds have a large photo-oxidant formation impact, anaerobic digestion might not be the best system to choose.

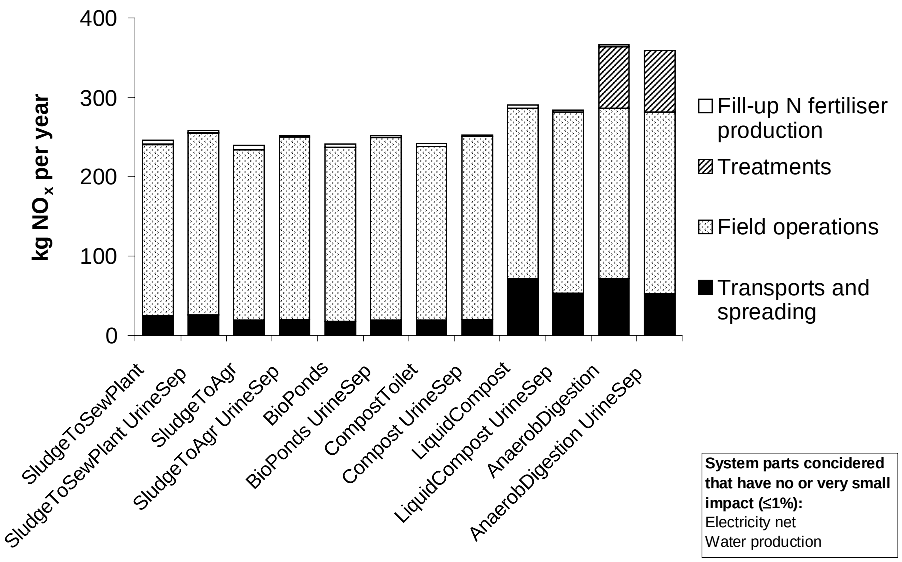

Figure 42. Emissions of nitrous oxides (NO_x, calculated as NO_2). Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(kg NO_x per year)*

Summary of results, organic waste management

Environmental impacts, energy consumption, primary energy consumption, amount N and P recycled, and additional yield of grain are summarised in Table 11. The figures given are total results per year. The environmental impacts and primary energy consumption are also shown in Figure 43 as relative values compared to the SludgeToSewPlant scenario. Heavy metal contamination of arable land is not included due to the complexity of those results.

To be remembered when studying the results is that the scenarios did not produce equal amount of utilities. They produced equal amounts of grain (with the help of fill-up N fertiliser), but while anaerobic digestion was a net producer of electricity, all others consumed this facility. Anaerobic digestion and liquid composting also produced heat. Furthermore, the blackwater-separating scenarios recycled more phosphorus, potassium, organic matter, and micro-plant-nutrients. This is probably important for long-term soil fertility but was not taken into account when grain production was calculated in the model, nor was the possibility of reusing water for irrigation in biopond systems accounted for. This should be included in future versions of the model to make fair comparisons.

Table 11. Summary of results of the organic waste management study. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(impact per year)* [1]

	Eutro-phic-ation	Global warm-ing	Acid-ifica-tion	Photo-oxid. POCP	Photo -oxid. NO_x	Energy consum ption	Primar y energy cons.	Re-used N-eff.	Re-used P-tot.	Addit-ional yield of grain
	kg O_2-equiv.	*kg CO_2-equiv.*	*kmol H^+*	*kg ethene-equiv.*	*kg NO_x*	*MJ*	*MJ*	*kg*	*kg*	*kg*
SludgeTo SewPlant	22000	17000	15	4.0	250	180000	230000	1300	470	30000
SludgeTo SewPlant UrineSep.	18000	17000	16	4.1	260	170000	220000	1400	490	32000
SludgeTo Agriculture	21000	17000	15	3.8	240	170000	220000	1300	480	30000
SludgeTo Agriculture UrineSep.	18000	17000	16	4.0	250	170000	220000	1500	500	32000
BioPonds	14000	16000	14	3.8	240	190000	260000	1300	480	30000
BioPonds UrineSep.	14000	16000	15	3.9	250	180000	250000	1500	490	33000
Compost Toilet	20000	19000	26	3.9	240	170000	220000	1300	510	30000
Compost UrineSep.	18000	17000	19	4.0	250	170000	220000	1500	510	33000
Liquid Compost	16000	19000	18	4.7	290	120000	350000	1400	510	31000
Liquid Compost UrineSep.	16000	18000	18	4.5	290	130000	300000	1400	510	32000
Anaerobic Digestion	13000	15000	28	4.5	370	230000	200000	1800	510	37000
Anaerobic Digestion UrineSep.	13000	14000	27	4.3	360	140000	90000	1800	510	38000
Anaerobic Dig. No Hygien-isation	13000	15000	28	4.5	370	100000	50000	1800	510	37000
Anaer.Dig. UrineSep. No Hygien-isation	13000	14000	27	4.3	360	50000	-20000	1800	510	38000

[1] All values given to two valid figures

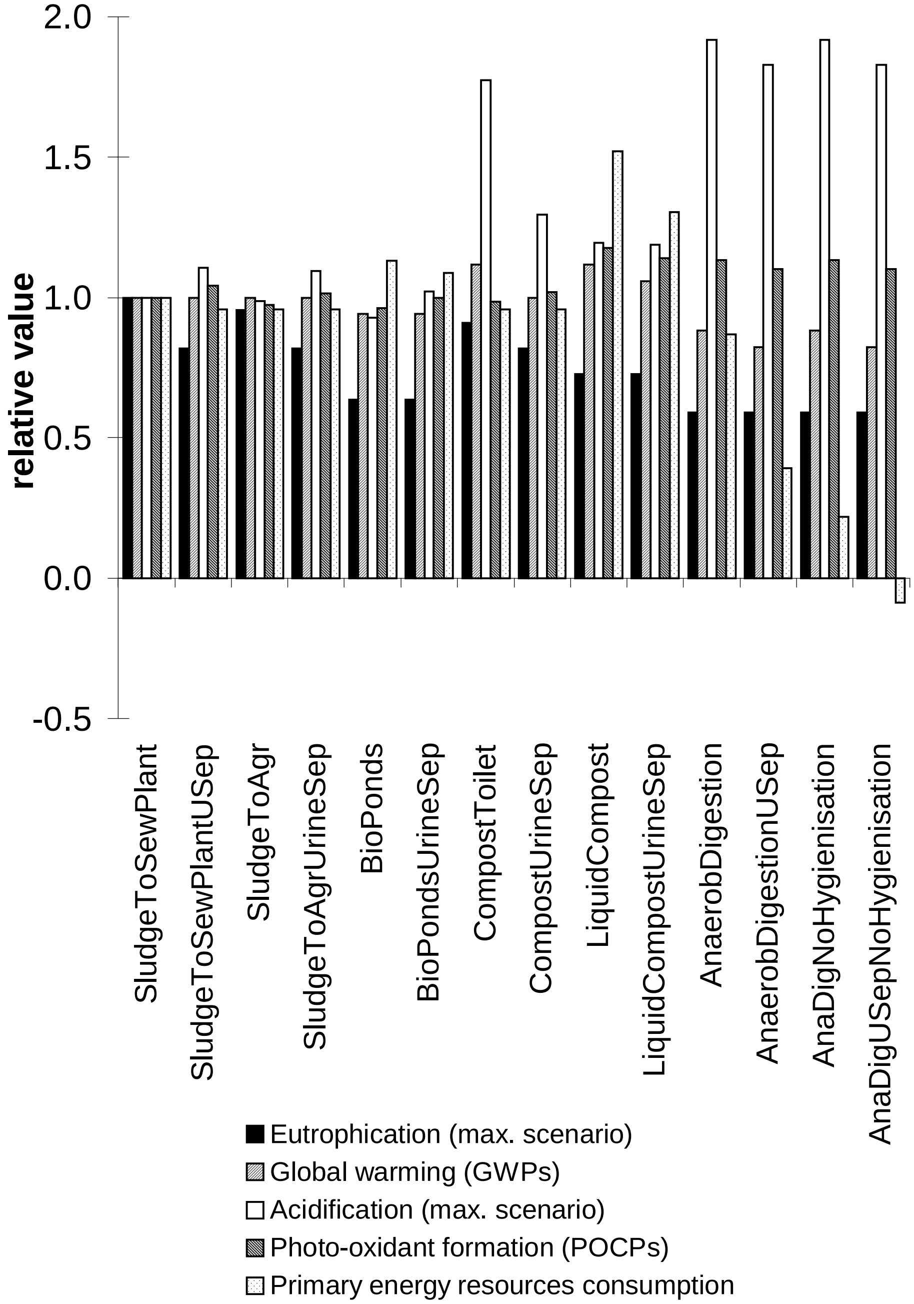

Figure 43. Environmental impacts, relative to the sludge–to-sewage-plant scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. *(relative values, 1=SludgeToSewPlant)*

Studying Figure 43 (or Table 11), it can be concluded that no scenario proved to be the optimal choice when all impact categories were taken into account. Thus, it is very important to find out which impact categories are most vital for the system or region of concern. For example, in many places the POCPs (photochemical ozone creation potentials) have a very small impact on the environment. In other places, acidification might not be considered as a problem.

Concerning the six scenarios (at the left of Figure 43) that had mixed-sewage-water systems, there were no big differences, except that the biopond scenarios had a lower eutrophication impact. On the other hand, the primary energy consumption was somewhat larger for the bioponds. Neither of the dry compost scenarios showed significantly different results, with the exception of a remarkably increased acidification impact due to ammonia emissions from the compost. By reduction of this emission, which is certainly possible, these scenarios would be on a level with the others to the left (Figure 43).

The results for the reactor-treatment scenarios, which had blackwater-sorting sewage systems, stood out some more. The liquid compost scenarios had a smaller eutrophicating impact, but larger impacts on all categories affected by fuel consumption. Pipe-transport of blackwater to treatments would probably reduce these impacts. Due to their relatively large electricity consumption, the liquid compost scenarios consumed more primary energy resources than all other scenarios. Anaerobic digestion scenarios produced the best results for eutrophication, global warming and primary energy consumption. However, for POCPs they had a somewhat larger impact and for acidification the largest impact of all systems. Concerning acidification, anaerobic digestion had double the impact compared to all other scenarios (except the dry compost scenarios). This was due to larger ammonia emission in connection with spreading and to sulphuric oxide emission when the biogas (methane) was combusted in a stationary engine.

Urine-separating toilet systems gave smaller environmental impacts in most cases. The largest effect was seen for primary energy consumption, where urine separation gave substantially lower consumption in the reactor-treatment scenarios. This was due to the decreased amount of water used in this kind of system. In the case of liquid composts, the smaller amount of water saved electricity used in the composting reactor. When it came to anaerobic digestion, the water saving decreased the heat consumption in the digestion reactor.

Sensitivity analyses; organic waste management

Electricity produced in oil-fired power plant

Swedish average electricity by no means represents a European average, since it is largely produced by equal parts of hydropower and nuclear power. Therefore, a sensitivity analysis where the scenarios were run with electricity produced in oil-fired power plants was also performed.

Global warming

The potential global warming impact was largely affected by the type of electricity production chosen in the scenarios using or producing much electricity (compare Figure 44 and Figure 38). Please note the differences in scale between the two diagrams. When electricity was produced from fossil oil, the CO_2-emission from this production was an important contributor to the greenhouse effect in the scenarios that used much electricity. The negative columns for the anaerobic digestion scenarios (Figure 44) reflect the emissions avoided due to the net production of electricity. The net result is thus the positive column subtracted by the absolute value of the negative column. The anaerobic digestion scenarios had almost no impact on global warming if the electricity produced substituted for oil-fired power plants. This assumption might be more adequate (than Swedish average electricity), if one considers this kind of electricity production as a marginal production, which it is likely to be, if it exists at all. The marginal production of electricity in Sweden has been from oil-fired power plants and to some extent imported electricity from Denmark and Germany, which have large proportions of coal-fired power plants.

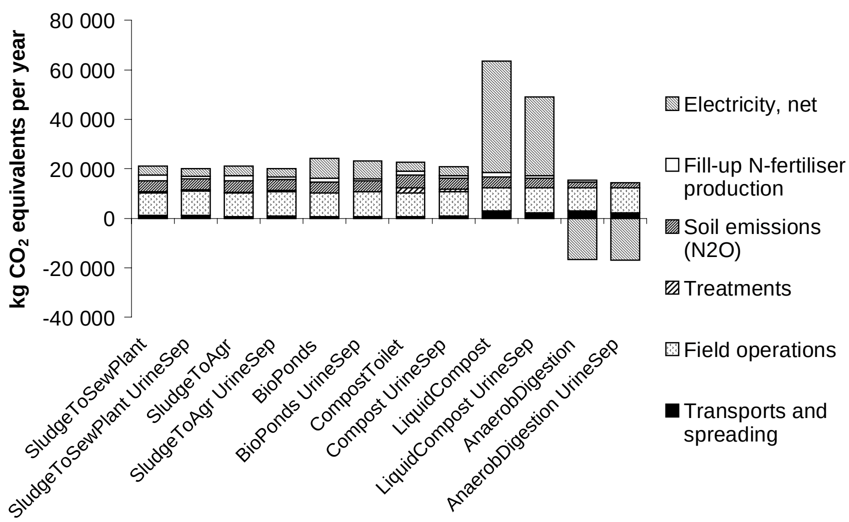

Figure 44. Potential global warming, 100 years scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain- production field operations plus additional production of nitrogen fertiliser. Electricity produced in oil-fired power plants. *(kg CO_2-equivalents per year)*

Acidification

When electricity was produced in oil-fired power plants (Figure 45), the emission of SO_x was also a major acidifying substance. For the two reactor-treatment systems (liquid compost and anaerobic digestion) the choice of primary energy source for electricity production was vital. Liquid composting scenarios, having about half the acidifying impact compared to digestion scenarios when electricity was produced as the Swedish average (Figure 39), doubled their impact when electricity was produced in oil-fired power plants. This made the net impact from digestion scenarios smaller than in the liquid compost scenarios.

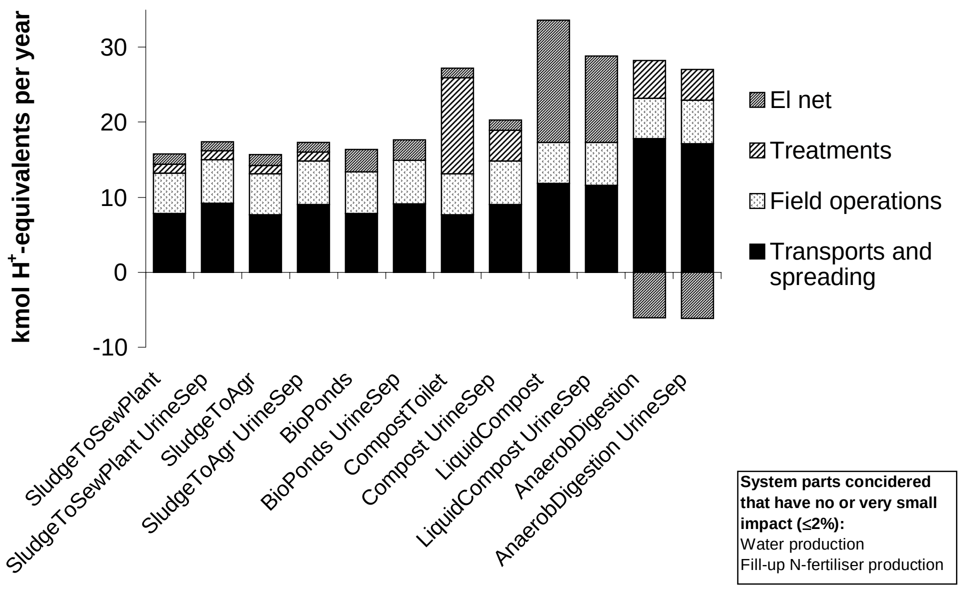

Figure 45. Acidification, maximum scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. Electricity produced in oil-fired power plants. *(kmol H^+-equivalents per year)*

As discussed earlier, the minimum acidification scenario could often be a more appropriate estimation of the acidifying impact in Sweden. The choice of oil-fired electricity changed the acidifying impact considerably (Figure 46 compared to Figure 40, please note the difference in scale). The large impact for the liquid compost scenarios emanated from emission of SO_x in the oil-fired electricity production.

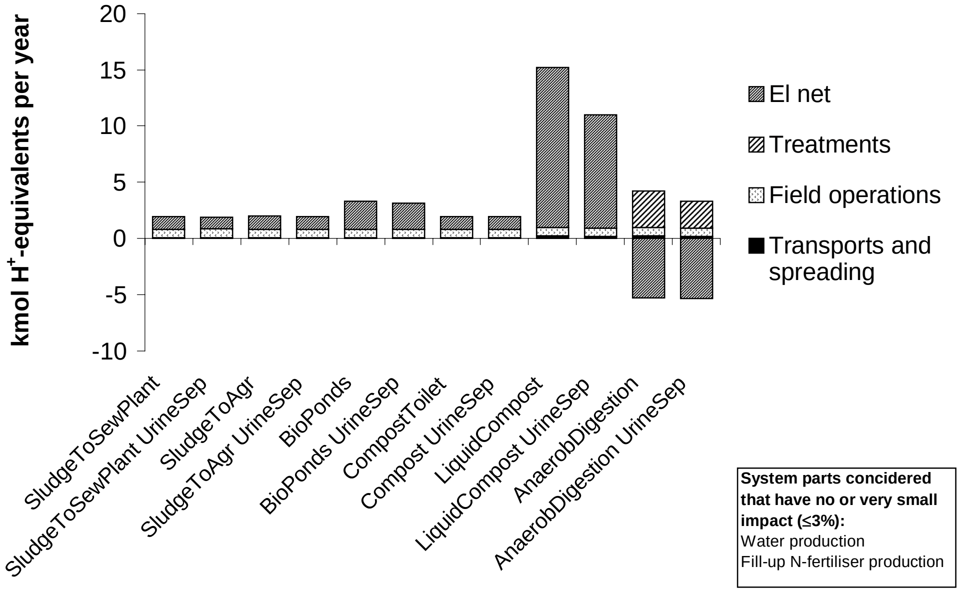

Figure 46. Acidification, minimum scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. Electricity produced in oil-fired power plants. *(kmol H^+-equivalents per year)*

Photo-oxidant formation

The choice of primary energy source for electricity production had vast consequences for POCPs (Figure 47 compared to Figure 41, please note the difference in scale), but not that large an implication where NO_x was concerned (Figure 48 compared to Figure 42). The exception was the liquid compost scenarios, which used much electricity. The very large POCP impact for electricity production when electricity was produced in oil-fired power plants was due to emission of NMHC, which had a very large photo-oxidant formation impact. Compared to this kind of electricity, electricity production from biogas seemed to be a very favourable alternative as regards the photo-oxidant formation impact.

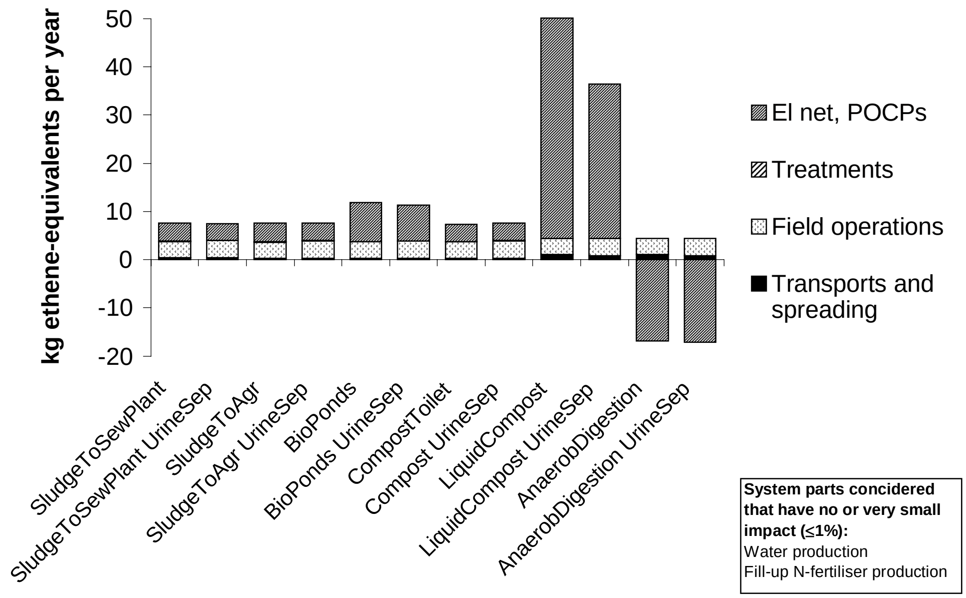

Figure 47. Photo-oxidant formation, using the concept of photochemical ozone creation potentials (POCPs). Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. Electricity produced in oil-fired power plant. *(kg ethene-equivalents per year)*

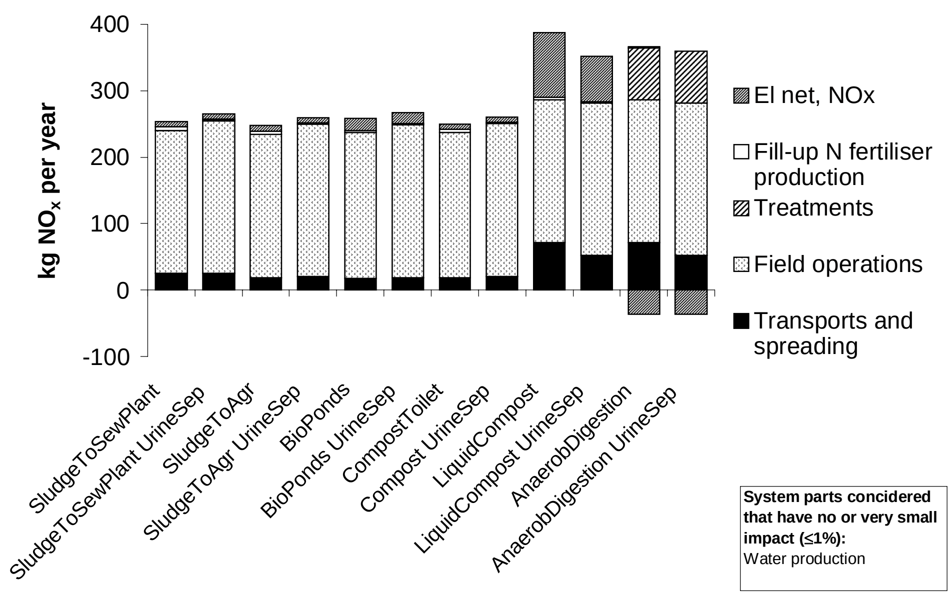

Figure 48. Emissions of nitrous oxides (NO_x, calculated as NO_2). Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. Electricity produced in oil-fired power plant. *(kg NO_x per year)*

Primary energy resources and eutrophication

The choice of primary energy source for electricity production did not have any great impact on primary energy consumption or on eutrophication (not shown).

Summary of results; oil-fired electricity production

The environmental impacts and primary energy consumption were shown earlier in Figure 43 (electricity produced as Swedish average) and below in Figure 49 (electricity produced in oil-fired power plants).

When the electricity consumed or produced in the reactor-treatment-systems was produced in oil-fired power plants, the results became clearly negative for liquid composting (although eutrophication was not affected). For anaerobic digestion, the outcome was the reverse. All impact categories, with exception of acidification, showed better figures since emissions from biogas electricity production were cleaner than the substituted (more polluting) oil-fired electricity production.

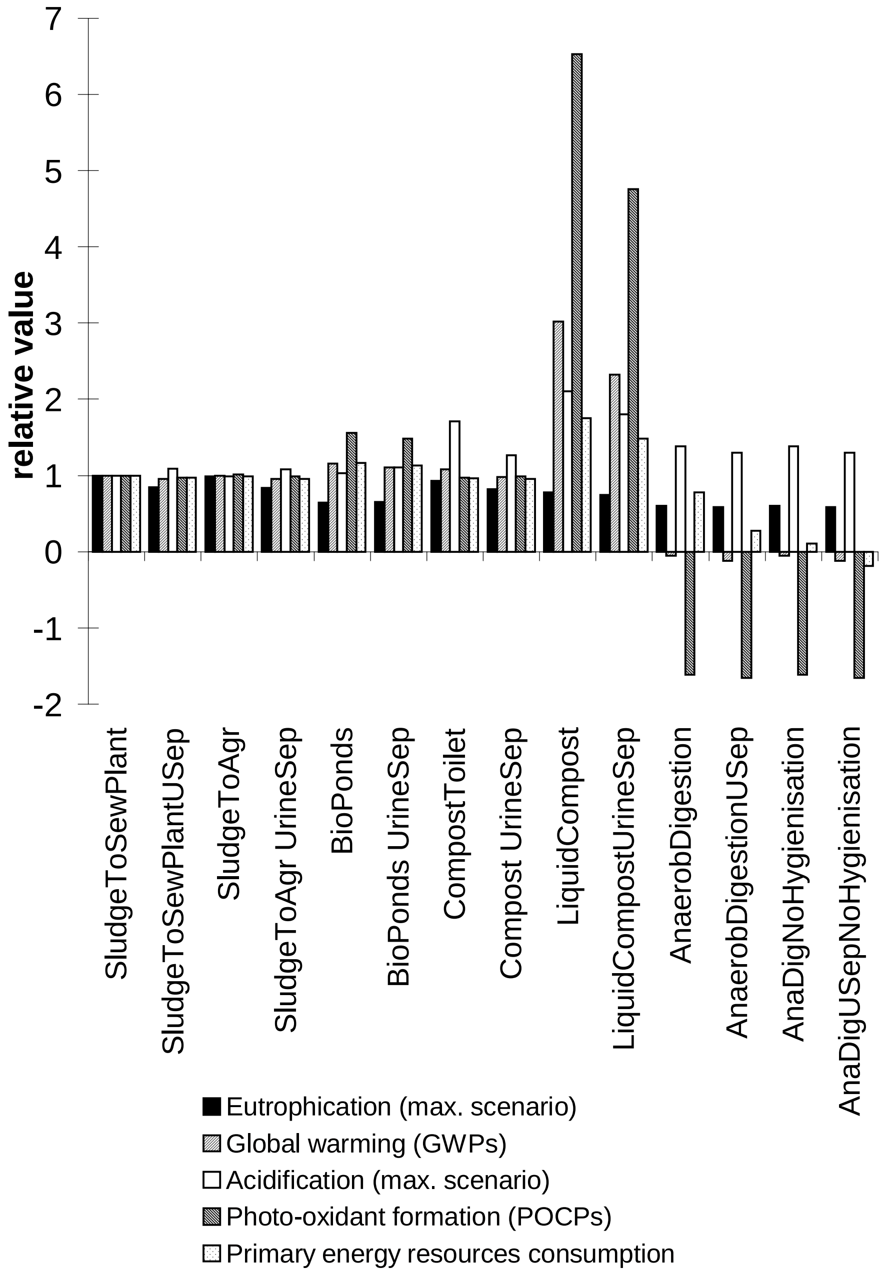

Figure 49. Environmental impacts, relative to sludge to sewage plant scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. Electricity produced in oil-fired power plant. *(relative values, 1=SludgeToSewPlant)*

Local transport distance set to 10 km

The choice of 2 km for all transport distances presumably affected the overall results. A set-up with the distance set to 10 km is therefore presented here. As can be seen in the following, it was mainly the liquid compost and anaerobic digestion scenarios which were affected. This was due to the higher transport requirements in these scenarios.

Energy turnover

The energy turnover when the transport distance was set to 10 km is shown in Figure 50. The equivalent for 2 km is shown in Figure 31 (please note the difference in scale). The longer transport distances resulted in increased energy consumption by about 30 % in the liquid compost scenarios, and by between 35 and 55 % in the anaerobic digestion scenarios (energy production not affected). For the other scenarios, the change was in the order of 10 %.

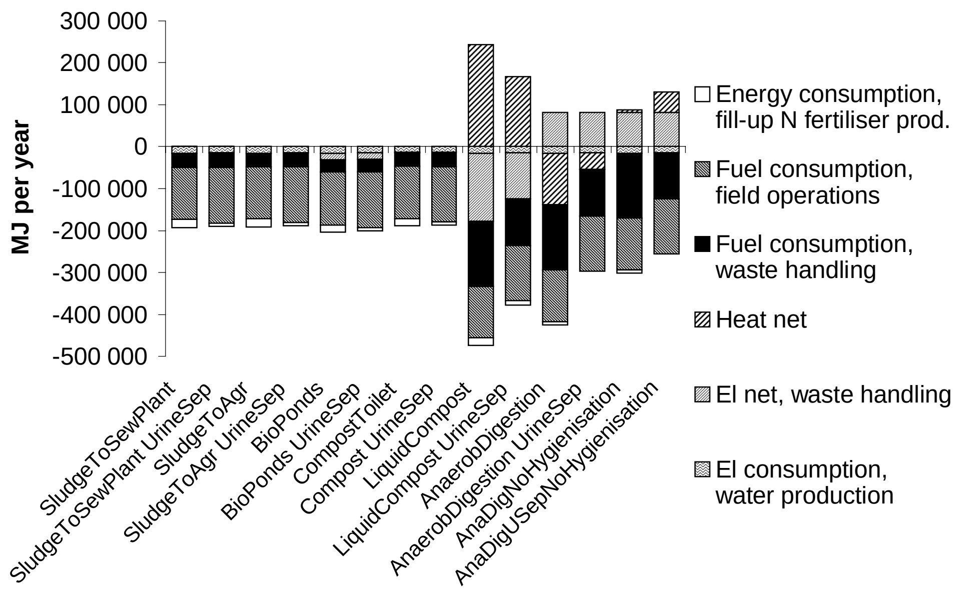

Figure 50. Energy turnover for treatment of wastewater and household waste from 32 households and manure from 15 dairy cows and their young stock, including water-production and grain-production field operations. Transport distance 10 km. *(MJ per year)*

Environmental impacts

The increase of transport distances from 2 km to 10 km gave the largest effect on photo-oxidant formation. The results for these environmental impacts are shown in Figure 51 and Figure 52. They can be compared to Figure 41 and Figure 42 (please note the difference in scales).

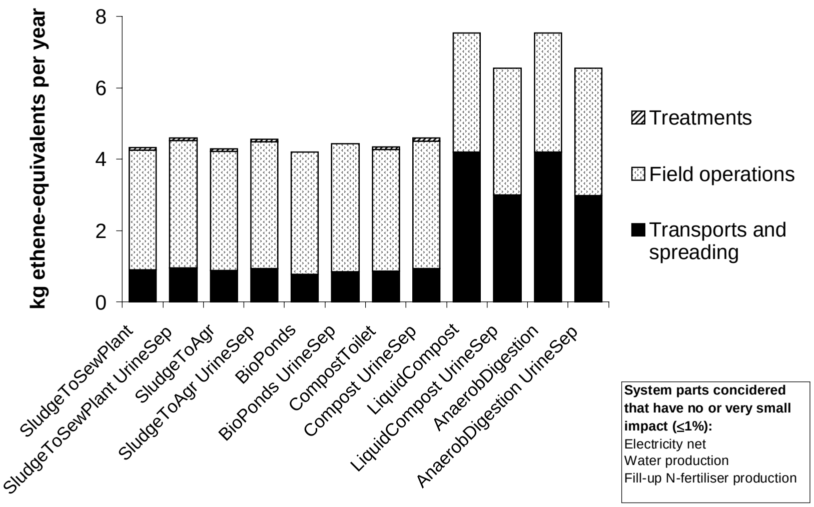

Figure 51. Photo-oxidant formation, using the concept of photochemical ozone creation potentials (POCPs). Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. Transport distance 10 km. *(kg ethene-equivalents per year)*

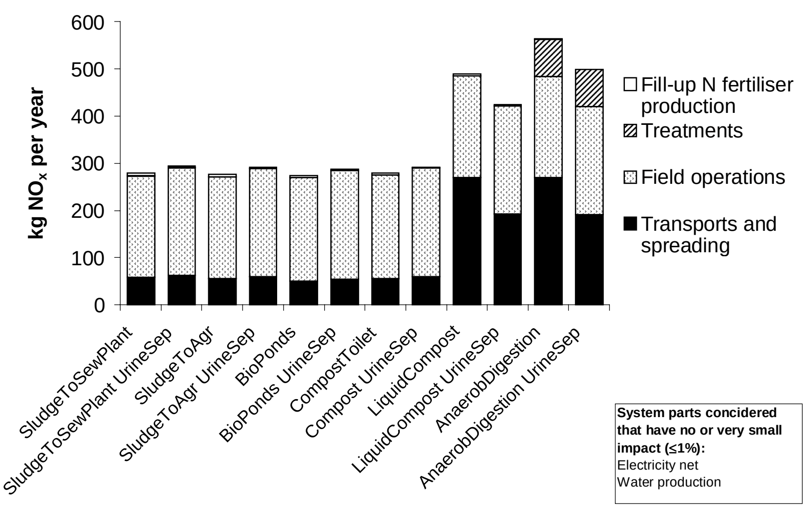

Figure 52. Emissions of nitrous oxides (NO_x, calculated as NO_2). Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain-production field operations plus additional production of nitrogen fertiliser. Transport distance 10 km. *(kg NO_x per year)*

Global warming was also largely affected (Figure 53). That diagram should be compared to Figure 38 (please note the difference in scale). Acidification was affected to a much lesser extent and eutrophication was almost unchanged.

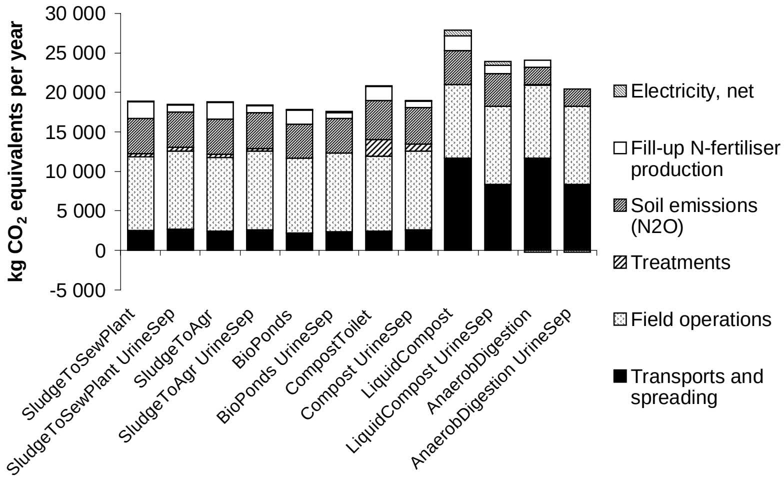

Figure 53. Potential global warming, 100 years scenario. Treatment of wastewater and household waste from 32 households, and manure from 15 dairy cows and their young stock; and grain production field operations plus additional production of nitrogen fertiliser. Transport distance 10 km. *(kg CO_2-equivalents per year)*

Bread production system

This chapter presents the bread production study. It starts with methodological issues and continues with the results and discussion.

System boundaries

The system under study included transport and processing of grain from the field to bread delivered at the consumer's door (Figure 54). The processes modelled were drying of grain, milling of grain, baking of bread (fermentation, oven, water heating etc.), clean water production, production of packaging for bread, handling of waste bread packaging and transport between all steps in the chain.

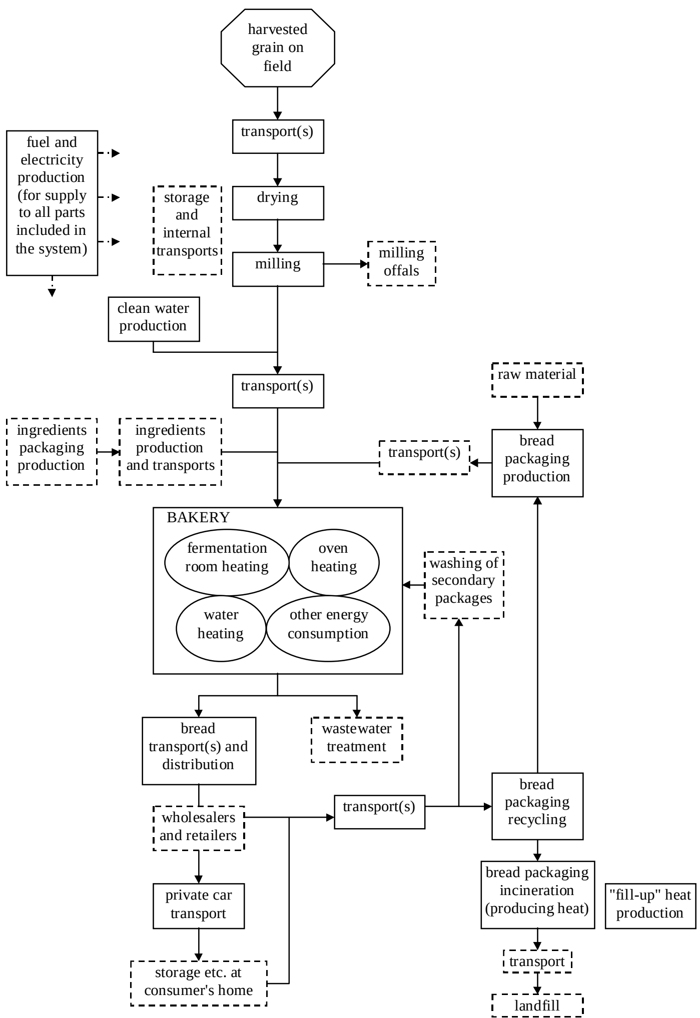

Figure 54. Flow chart of the bread production system. Dotted blocks not included in the study.

Agriculture and cropping of grain were not included in this analysis, neither were the consumer phase nor its storage. These are of importance but were assumed not to change between the different scenarios and were thus not needed for the comparison. In addition, the agricultural parts were dealt with in the organic waste management study, and thus included in the example of a cyclic small-scale food system. The retailer level was also omitted since for bread it was assumed to have little impact on energy and resources consumption and since it would anyhow be about equal for all scenarios. Other parts of the system were omitted from this study because they were assumed to be negligible concerning energy consumption and environmental impact in comparison to those included. These were storage of grain and flour, the fate of milling offals, production of additives (sugar, yeast, spices, etc.), packaging for additives, wholesalers, cleaning of plastic boxes and wastewater treatment. With regard to the packaging which was included, the transportation of such packaging from manufacturer to bakery was not included.

The model outputs are yearly amounts of emissions and energy consumption, but all results were recalculated to emissions and energy consumption per kg bread produced. This was chosen to get more easy-to-understand results. To obtain the results for a 100-person settlement (the size used in the organic waste management study) the per-kg-bread results have to be multiplied by 3500; 35 kg bread per person and year in Sweden (Becker, 1992) times 100 persons. Furthermore, in order to get the scenarios to produce equal amounts of heat (in the incineration of packaging wastes) "fill-up" heat production was included in the system.

Assumptions

All grain used was assumed to be produced in Sweden, and thus no difference in cropping environmental effects was accounted for. Imported grain is sometimes used to get a higher protein (gluten) content in the flour, but there seemed to be no difference between scales in this aspect.

No losses of grain or bread were included. This might be a source of errors since there probably are losses, especially of bread in later stages of the chain. The differences between scales are, however, probably not that large.

Electricity used was assumed to be produced as Swedish average electricity, i.e. about equal parts of hydropower and nuclear power with minor parts produced from biofuels, wind, oil, and coal.

When plastic boxes were used for secondary packaging, it was assumed that they were transported back to the bakery by returning lorries, and thus did not create any extra transport. The emissions and energy use for cleaning of them is negligible (Andersson, 1998a) and was thus not included.

The energy consumption was calculated as electrical and thermal energy. Thermal energy consumption was the sum of fossil fuels (mainly oil and gas), bio-fuels and internal heat in packaging materials minus the heat recovered by incineration of packaging materials.

Scenarios

Four main scenarios were presented. Two were large-scale systems that produced bread distributed over large distances with long-distance transport plus local distribution, and two were small-scale systems producing bread for a local/regional market. Both large-scale and small-scale systems were represented by one "irrational" and one "rational" scenario. *Irrational* refers to both the logistical planning of production and to the use of old, non-energy-optimised technology. The logistical planning in the bakery is important because if e.g. pre-heating time and shifts between different sorts of bread in the oven are not minimised, the time for oven heating gets longer and, thus, energy consumption gets larger. *Rational* refers to systems where both logistical planning and process control were optimised in order to minimise energy consumption and environmental impacts. Additionally, a fifth scenario was included. It was the small-scale rational scenario complemented with the private-car transport used in the other scenarios.

The large-scale scenarios represent realistic "extremes" considering energy use that seem to exist in Sweden (and probably elsewhere). Assumptions about different transport types were intended to represent common situations in Sweden. All transport was carried out by diesel-fuelled vehicles unless otherwise stated. Plastic bags were used as primary packaging for the bread in all scenarios.

1. *Irrational large-scale*, used oil as fuel in an indirect heated oven and steam boiler. In many ways, it resembled a real bakery system described as Industry 1 in Andersson (1998a). Drying of grain was done in an oil-fired indirect heated drier. The secondary packaging for bread distribution was cardboard boxes that were used only once. Grain was transported 30 km by tractor and wagon (same in all scenarios) to drying and milling plants and then 210 km by truck and trailer to the bakery. Bread was first transported 260 km by truck (inter-regional), then locally distributed to shops by truck (4 tonnes full load, 350 km round-trip), and finally transported 5 km by private petrol-driven car to the consumer's home.

2. *Rational large-scale*, was in many ways similar to Industry 2 in Andersson (1998a). It used electricity for heating of oven, steam boiler, and fermentation room. The grain was dried in a gas-fuelled direct-heated drier. The secondary packaging for bread distribution was plastic boxes that were used 500 times. Transport assumptions were equal to those in the irrational large-scale scenario.

For the small-scale scenarios, the intention was to present one environmentally optimised system and one system that developed following other criteria. Both had short transport distances in all parts of the chain. The secondary packaging used was plastic boxes.

3. *Irrational small-scale*, had a bakery that used an oil-fired indirect heated oven and an oil-fired steam-boiler. The grain was transported 30 km by tractor and wagon to an oil-fired indirect heated drying plant, which was co-located with the milling plant and the bakery. The bread was distributed locally to the shops by a diesel-fuelled pick-up or van (600 kg full load, 100 km round-trip) and then, as in the large-scale scenarios, transported to the consumer's home by private (petrol-driven) car.

4. *Rational small-scale*, used electricity for all heating in the energy optimised bakery. The other big difference between these two scenarios was the home-transport of bread. Here it was done by foot, by bicycle, or by home-distribution carried out by the bakery. In the latter case, the retailer level was not needed and it was assumed that the home-distribution could be carried out in an energy efficient manner, thus substituting for the local distribution from bakery to retailers. All other transport assumptions were equal to those in the irrational small-scale scenario. Other differences between the irrational and rational small-scale scenarios were the grain-drier, which here was indirectly heated by biofuels, and the mill, which used an energy-saving mechanical internal transport system.

5. *Rational small-scale + car transport*, was equal to the fourth scenario but complemented with vehicle-transport to the consumer's home by private (petrol-driven) car as used in the first three scenarios.

To choose electricity as major energy carrier in the rational scenarios might not be self-evident. As the Swedish average electricity is largely produced by hydropower and nuclear power, which is quite "clean" in comparison to electricity produced from fossil coal and oil, the choice was logical. However, land use, storage of nuclear waste, etc. were not taken into account when this choice was made. In order to evaluate the choice, sensitivity analyses where electricity was produced in oil-fired power plants and rational scenarios having gas-fuelled ovens are presented at the end of this chapter.

Models

The general structure of the model is shown in Figure 55. Each block in the diagram represents a sub-model that work independently of the others. The grain is defined at the top as a certain amount of dry matter and water. Then the transformation of incoming material (where appropriate), energy consumption, air emissions from fuel consumption, and possible water emissions are calculated in each sub-model. Emissions for electricity production are calculated later in a general spreadsheet programme (MS Excel).

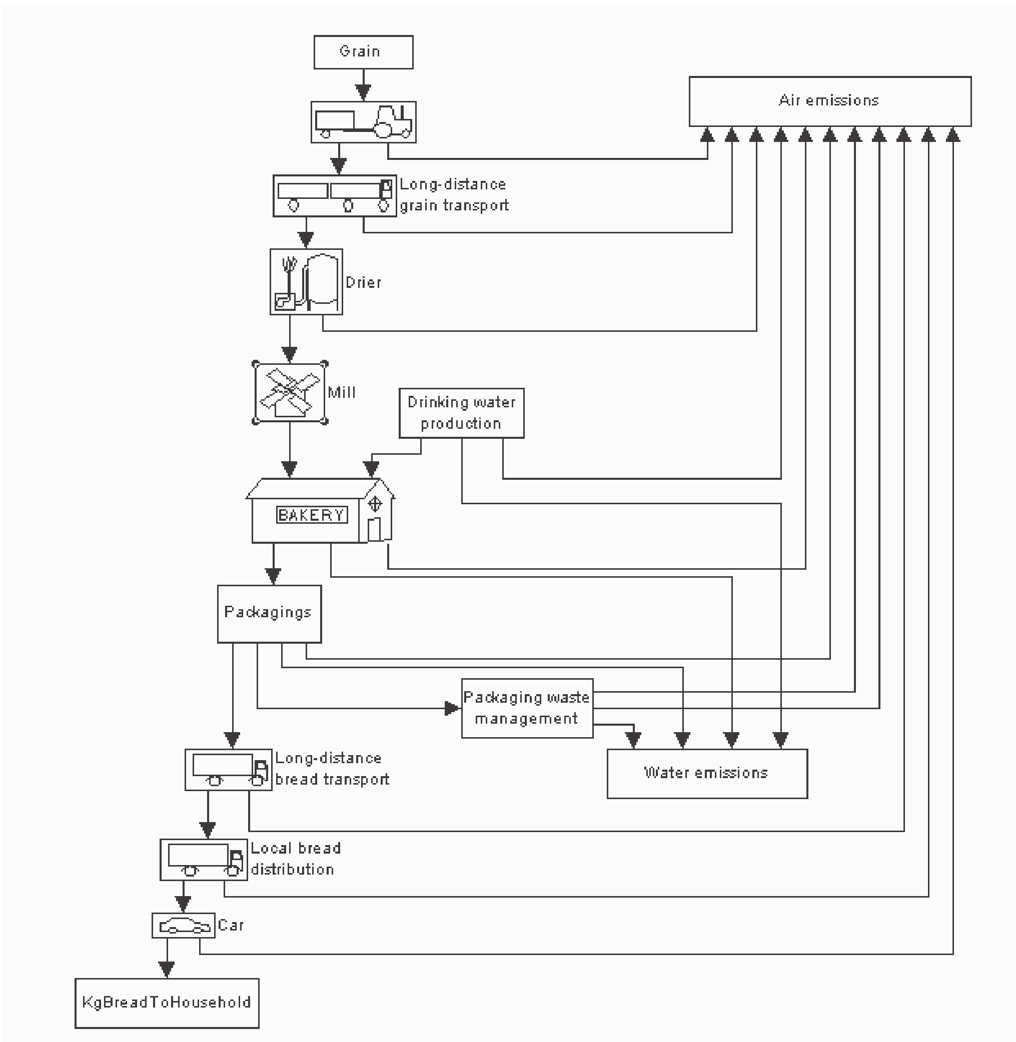

Figure 55. General structure of the bread production system model.

Clean water production

Water of drinking quality is used in the bakery as an ingredient in bread and for cleaning. Some water is also used for steaming but most of the condensed steam is re-circulated so a very small amount is needed for this purpose. The model is described in the chapter "Organic waste management system".

Grain processing

Grain drier

In Sweden, grain usually needs to be dried before storage, but the initial moisture content varies a lot between years and between regions. Thus, the model had to be flexible in this aspect. The grain drier model (Figure 56) calculates the amount of water vaporised from chosen values of moisture content wet base (m.c.) for incoming and dried grain. In this study, grain with 20 % m.c. was dried to 14 % m.c. The amount of water in dried grain is calculated from the amount of dry matter (which was assumed to remain unchanged) and the desired m.c. The energy consumption is calculated by multiplication of factors for heating energy consumption and for electricity consumption for fan propelling, with a calculated amount of vaporised water. Finally, air emissions due to fuel consumption are calculated. The Demux-block split the 43-position vector into smaller vectors and/or scalars. The Mux-block aggregates them to a 43-position vector again.

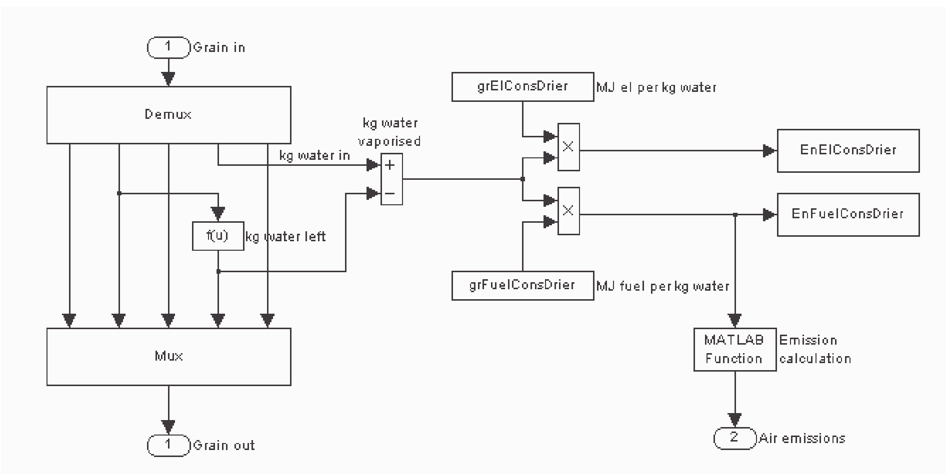

Figure 56. Grain drier sub-model.

In practise, the heat consumption is dependent on initial m.c. The higher the initial m.c., the lower the energy consumption per kg water vaporised. I used 5.4 MJ_{heat}/kg water, which is an approximate average for drying of grain with moderate m.c. in indirect heated driers (Regnér, 1998).

Direct heated driers using e.g. fossil gas as fuel have 15-30 % lower energy consumption (Regnér, 1987). When used in this study, they were assumed to have a 20 % lower energy demand, i.e. 4.3 MJ_{heat}/kg water. The electricity demand in all cases was assumed to be 0.3 MJ_{el}/kg water (Statens Maskinprovningar, 1987). The model finally calculates the air emissions per MJ fuel used. For more details and references, see Appendix 5.

Flour-mill

Energy consumption for milling of grain varies a lot depending on e.g. variety and moisture content of the grain, types of flour produced (rate of extraction), and type of internal transportation (mechanical or pneumatic) in the mill (Jakubczyk et al., 1987). The internal transportation is nowadays often done in pneumatic systems, which are cleaner but more energy-demanding than mechanical systems. Different sources give energy consumption between 0.1 and 0.45 MJ_{el}/kg flour depending on transport system and flour extraction rate (Jakubczyk et al., 1987; Andersson, 1998a). The lowest figures are valid for systems with mechanical transportation systems and high extraction rates. The structure of the mill model is shown in Figure 57.

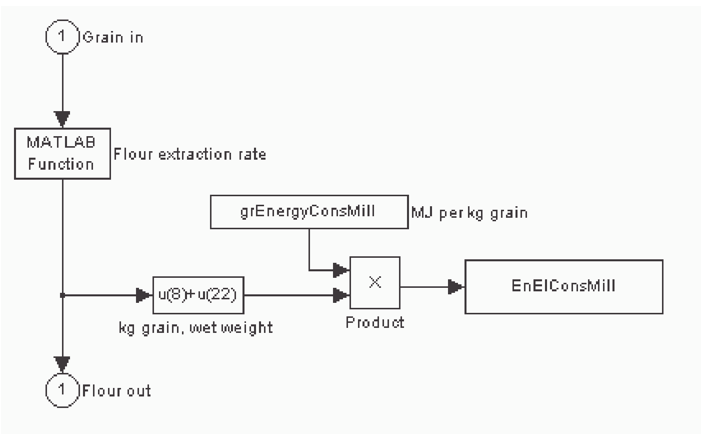

Figure 57. Grain mill sub-model.

The electricity consumption was set to 0.36 MJ_{el}/kg flour (wet weight) in both large-scale scenarios and in the irrational small-scale scenario. For the rational small-scale scenario I assumed the energy consumption to be considerably smaller, about 0.15 MJ_{el}/kg flour. This level is among the lower range of those presented by Jakubczyk et al. (1987), and is also indicated to be valid for the small-scale bakery system (including mill) studied in Andersson (1998). The difference in the latter case is probably a result of above all a mechanical transportation system in combination with higher extraction rate and maybe a larger proportion of rye (which is easier to grind than wheat).

For practical reasons, a medium-range (80 %) flour extraction value was chosen for all scenarios. This did not have any large implications since the milling offals were not included in further calculations, though they might be used both as food for humans and feed for animals. This simplification should be of minor importance since the flour is the main product and milling is a minor energy-consuming part in the whole system. Of same reason, I chose to allocate all milling energy consumption to the flour.

Bakery

The bakery model calculates the energy consumption in different parts of the bakery process (Figure 58). Different literature sources have aggregated (or have measured) data in different ways but most of them give at least oven heating and total energy consumption per kg bread produced. Therefore all calculations are performed per kg bread produced and thus the amount of bread produced per kg flour first has to be calculated. This varies a lot between different bakeries (and probably between different kinds of bread). In this study, it is chosen to use the value 1.4 kg bread per kg flour in all scenarios. Furthermore, it was assumed that all the added weight was water, i.e. additives were neglected. Kent & Evers (1994) stated that one can get 145 kg bread out of 100 kg flour and that only 4 kg additives need to be added, i.e. about 3 % of the bread weight. In the environmental analysis of a medium-sized Swedish bakery (Nilsson, 1996) it is reported that the additives stand for about 13 % of the total bread weight. To leave the additives out might thus be a source of error in the calculations but I estimated it to be small.

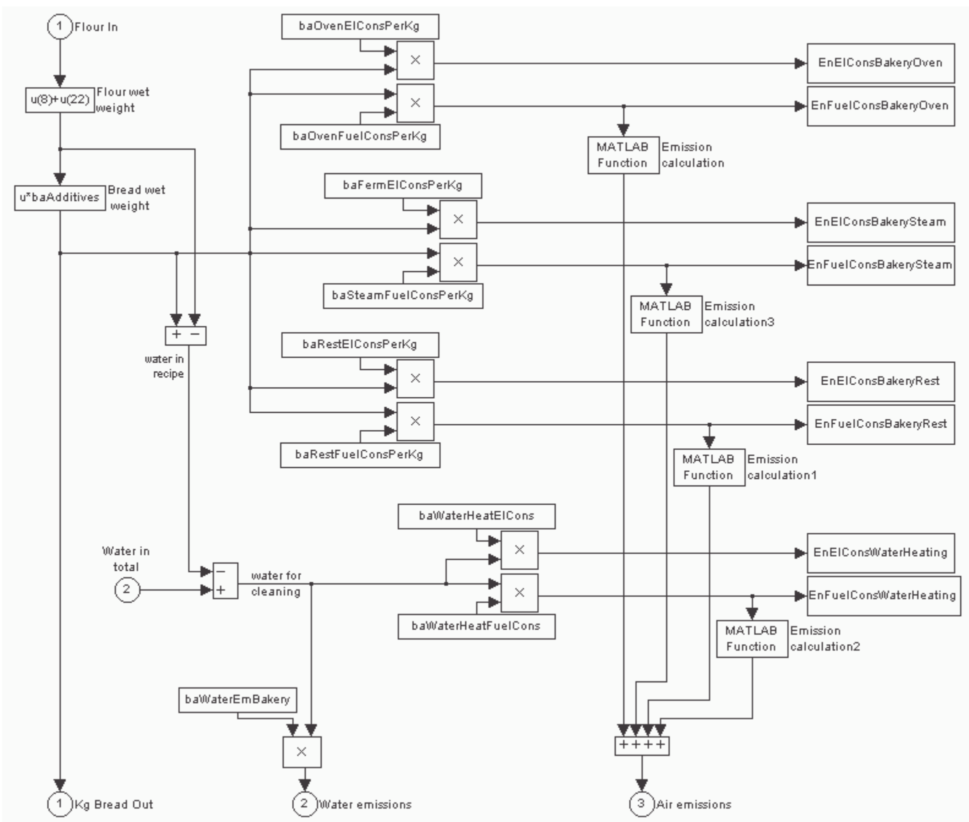

Figure 58. Bakery sub-model.

The model makes calculations for oven heating, fermentation room heating, steam production, water heating, and a remaining part separately. For these, it is possible to use electricity and/or fuel as energy source. For air emission calculation, it is possible to choose different fuels for heating of different processes. It is possible to calculate the water emissions in the model but since they are of very small proportions (Andersson, 1998b; Nilsson, 1998) they were not included in this study. However, the amount water consumed was calculated for use in the water-heating energy consumption sub-model. Energy consumption for water added to the dough is assumed to be included in the other parts. For more details see Appendix 6.

Packaging

Andersson (1998b) found that the packaging used for the bread is one of the major energy-consuming parts of the system. The data concerning packaging used in Anderson's Life Cycle Analysis are published by Arnkvist (1997), who has closely investigated the use of packaging in bread production.

Included in the model calculations were primary packaging (paper or plastic bags) for bread and secondary packaging (corrugated cardboard or plastic boxes) used for distribution of bread. The dominating bread primary packaging in Sweden is plastic bags made of low-density polyethylene (LDPE). Paper bags are usually only used by small bakeries for bread sold directly to customers (without distribution to retailer). Bakeries that distribute their bread over large areas (long distances) use non-returnable corrugated cardboard secondary packaging, while many bakeries with local or regional distribution areas use plastic boxes made of high-density polyethylene (HDPE). They are brought back to the bakery, where they are cleaned, and used again about 500 times (Arnkvist, 1997). The packaging for ingredients is in most cases a small part of the total (Arnkvist, 1997) and was thus omitted. This assumption might lead to an under-estimation of the environmental impact from packaging, especially for the small-scale scenarios, but it should be of no vital importance.

The general structure of the packaging sub-model is shown in Figure 59. At the top, the amount of bread is introduced. The choice of primary packaging type is done by definition of the constants baPlasticBagChoice and baPaperBagChoice to either zero or one, or parts of one, i.e. either plastic or paper bags, or a combination of the two. The choice of secondary packaging is performed in a parallel manner. When plastic bags are chosen, plastic clips (polystyrene) are also included.

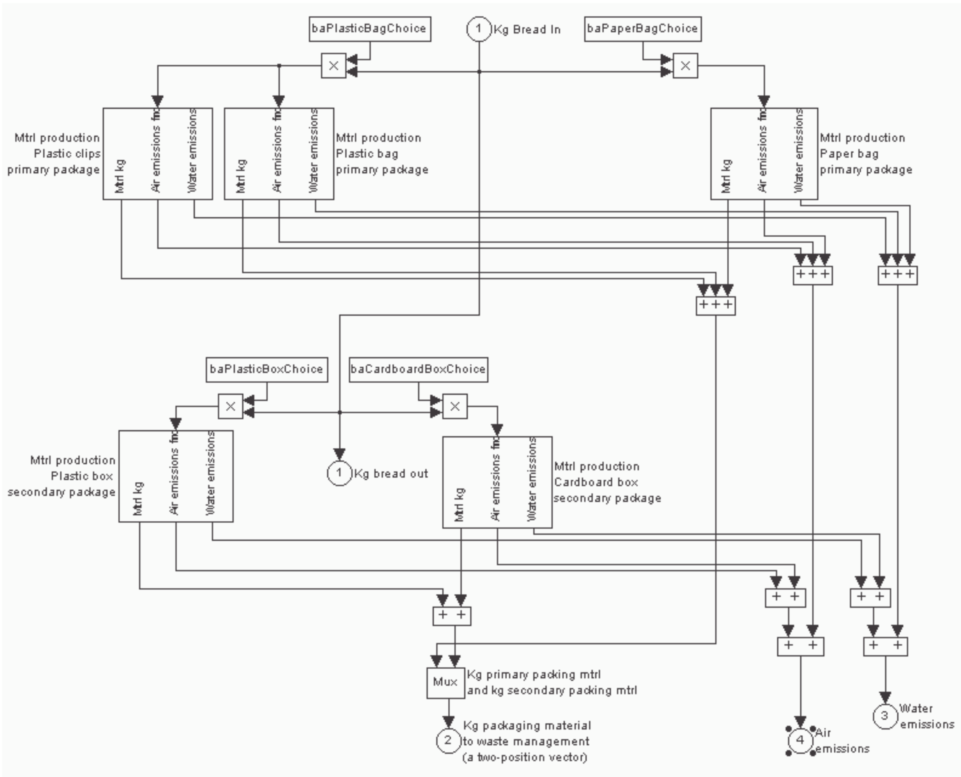

Figure 59. The sub-model for choices of primary and secondary packaging.

The air and water emissions and energy consumption due to production of the different materials are calculated in the separate parts of the model. Figure 60 shows the plastic bag production sub-sub-model. The structure of the others is the same.

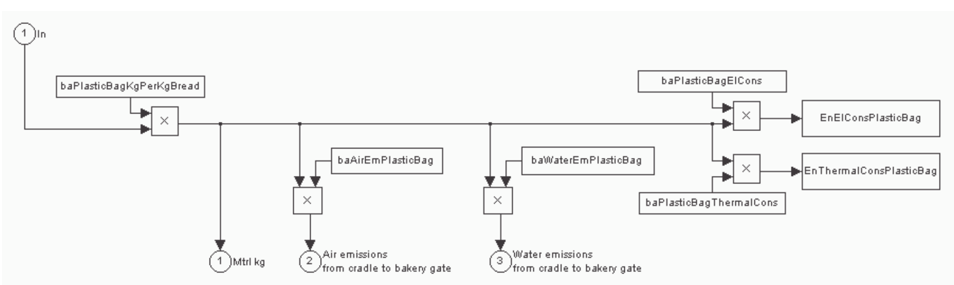

Figure 60. Plastic bag production sub-model.

Lifecycle data are used, i.e. all emissions from raw-oil extraction or timber felling to the packaging entering the bakery door are included. Data for the production of plastics are published by Boustead (1993, 1997). Data for paper and cardboard come from Habersatter et al. (1998). For more details see Appendix 6. The data for packaging are valid for Central European manufacturing. Compared to Swedish conditions, these data probably give more environmental emissions from energy consumption, since coal and other fossil fuels are more frequently used as energy carriers. However, this should not be of great importance.

Packaging material waste handling

After use, it is assumed that both primary and secondary packaging are either incinerated or recycled. However, in this study, all scenarios had recycling of packaging materials. Incineration was still needed since the materials could not be recycled more than a certain amount of times. In practice, this is reflected as a loss of material in each recycling step. Some of this material is in reality put on landfill, but that was not modelled since landfilling of incinerable wastes is not preferable from an energy utilisation point of view (Sonesson et al., 1997). Furthermore, landfilling of incinerable wastes will be banned in the European Union in a few years time. The incineration plants are often connected to district heating systems, thus replacing heat produced from other energy carriers. When the material is recycled, it replaces virgin fibres in the production of paper or cardboard and fossil oil in the production of plastics. Concerning plastics, the recycled material is not used for same type of products as in its first lifecycle, but it still replaces virgin oil. Figure 61 shows the structure of the recycling model. First, the choice of recycling or incineration and type of packaging is defined.

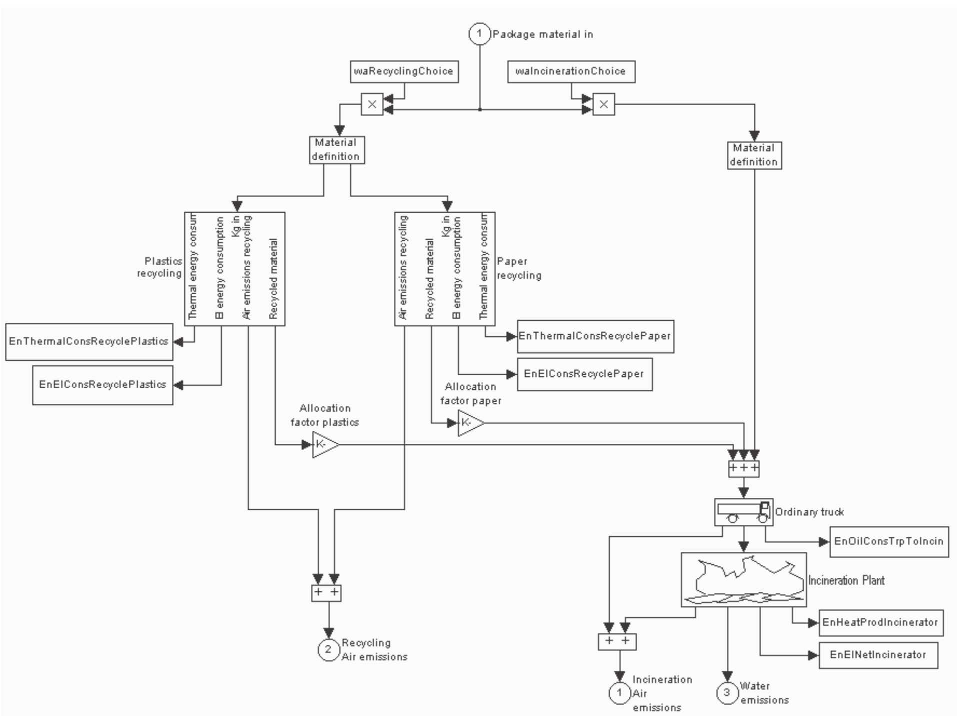

Figure 61. General structure of the packaging waste handling sub-model.

When incineration is chosen, energy consumption and production, and air and water emissions due to transportation and incineration of all material are calculated. The incineration sub-model is described in detail in Mingarini (1996). When recycling is chosen the energy consumption and air emissions due to transportation and processes created by the recycling are calculated. The allocation factors for plastics and paper make it possible to decide how many times the material can be assumed to be recycled before it goes to incineration. Here, it was assumed that plastics were recycled once, while paper products were recycled four times. This choice had no large impact on the results.

The paper recycling process sub-model is shown in Figure 62. The sub-model for plastics is equivalent. Data for recycling of packaging materials were as given by Tillman (1991). To get the share of energy consumption and emissions originating from the recycling, I simply took the differences between recycling and landfilling scenarios for paper and plastics respectively as they are given in that reference, i.e. the transport was included with assumptions about distances etc. made in that study. For all plastics, figures for LDPE were used, thus assuming that LDPE, HDPE and PS were equivalent. For paper and cardboard, data for cardboard were used, thus assuming that the two were equivalent. Both these assumptions made the results less accurate but any possible errors that could arise should be small compared to the energy consumption and environmental impacts for the whole system. To obtain total energy consumption and air emissions, the data are multiplied by the actual weight of material. For more details see Appendix 7.

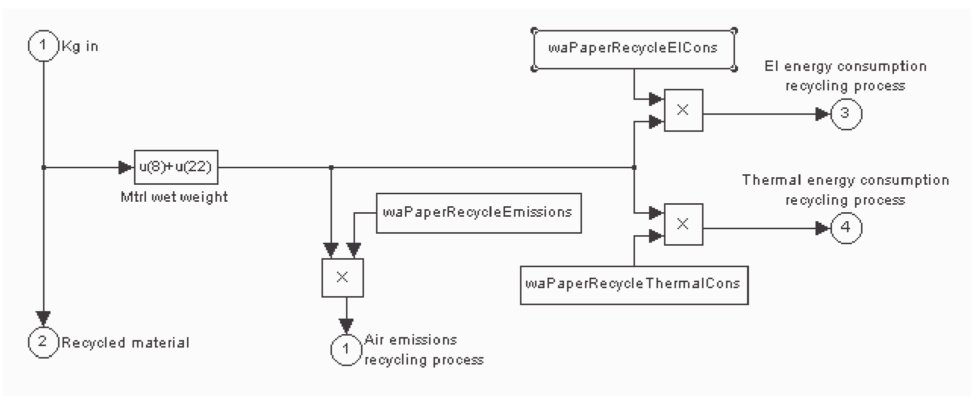

Figure 62. Paper recycling sub-model.

Transport

In the model, the grain, and later the bread, are transported by vehicle between five different points in the chain from field to consumer in the large-scale system. In reality there might be more transport steps but since I did not include loading and unloading in the analysis some simplifications have been possible. In the small-scale system, this could be reduced to two or three steps.

Four different types of vehicles were used; agricultural tractors with trailers, lorries, pick-ups or vans and private cars. The models were the same as those used in the organic waste study presented under "Transport" in the chapter "Organic waste management system". All vehicles except private cars were diesel-fuelled. For more details see Appendix 4.

Transport of grain from field to drier

Tractor with wagon was used for transport from field to grain drier. The one-way distance was assumed to be 30 km, and to be the same in all scenarios. In practice this transport may be divided in two, first a short transport from the field to an on-farm located drier and thereafter transport to a central storage site.

Long-distance grain transport

Truck and trailer transports the grain to the mill (in the large-scale scenarios). The one-way distance was assumed to be 210 km (as given by Andersson 1998a). The vehicle was assumed to carry a full load (30 tonnes) out and to return empty. Fuel consumption was 5 litres/10km with a full load and 3 litres/10km empty. In practice this transport might also consist of two parts, grain from storage (on-farm or central) to the mill and then transport of the flour to the bakery. However, this is of no importance in the systems perspective of the study.

Inter-regional bread transport

In the first step of the large-scale systems, the bread was assumed to be transported by truck 260 km one-way. This is the average distance for a large Swedish industrial bakery reported by Andersson (1998a). Fuel consumption was assumed to be 4 litres/10 km with a full load (13.3 tonnes) and 2 litres/10km when empty. The assumed load-factor was 70 %.

Local bread distribution

The local distribution of bread can be performed in different ways and is highly dependent on local and regional conditions. In the large-scale scenarios I assumed that the round-trip distance was 350 km, which is equivalent to the distance given by Andersson (1998a). The fuel consumption was set to 5 litres/10km and the max-load to 4 tonnes. For the small-scale bakery, the distribution step was done using a smaller pick-up with 600 kg bread loaded when it leaves the bakery. Its fuel consumption was assumed to be 1.1 litres/10km and the round-trip was 100 km. These load and distance data were chosen from existing tours which were assumed to be relevant (Nilsson, 1996). The fuel consumption data were as yet unpublished data from Girma Gebresenbet, Dept. of Agr. Engineering, SLU (pers. comm.).

Private car transport

The transportation from the shop to the consumer's home was assumed to be done by petrol-driven private car in scenarios 1, 2, 3, and 5. To get the environmental impact and energy consumption of this transport, some assumptions had to be made. The one-way distance driven was 5 km and the trip was made exclusively for shopping. Two kg of bread were bought each time, which is the approximate weekly consumption for a three-person family (Becker, 1992). Bread is about 6 % of our average food consumption expenditure (SCB, 1999). Thus, 6 % of the transport was allocated to the bread. Fuel consumption (1.23 litres/10km) and emission factors are data from Eriksson et al. (1995).

Results and discussion of the bread production system

The results for the bread production study are presented and discussed first in impact category by category. At the end of the chapter, there follows a summary which compares the impacts relative to the large-scale irrational scenario.

Energy consumption

Processes and transport are large parts of the energy consumption, as can be seen in Figure 63. Choice of energy source for heating of bakery oven and fermentation room affect the relationship between electrical and thermal energy use but not the total energy consumption to any large extent. The difference in process energy consumption between rational and irrational scenarios was chosen to represent different levels of energy optimised systems. Anyhow, in practice, electrical heating of the bakery oven and fermentation room might reduce the energy consumption somewhat because direct heating can be used.

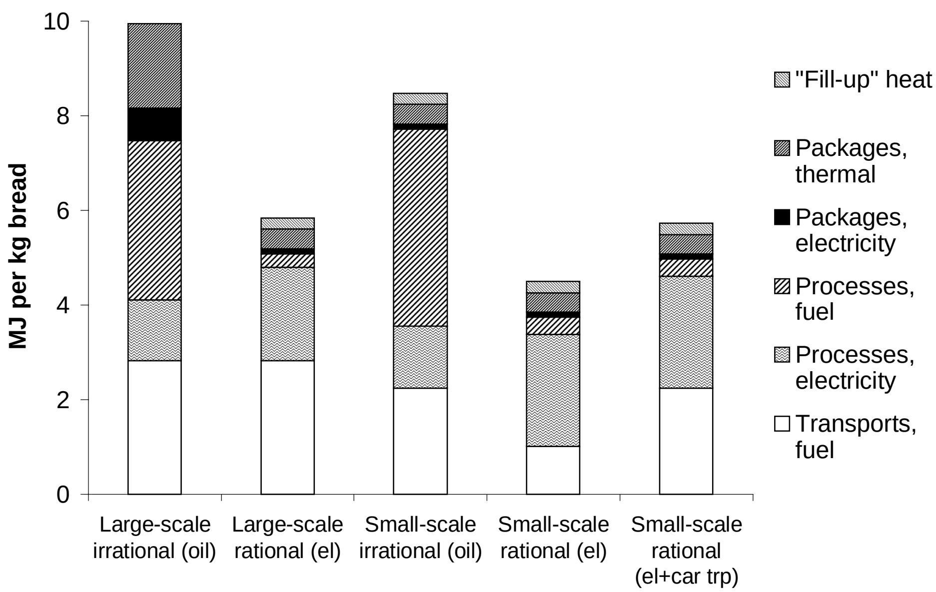

Figure 63. Electrical and thermal energy consumption due to transport, processes, and packages. Transport was that performed within the system; i.e. transport in connection with e.g. package waste handling were not included here. Processes included grain drying, grain milling, bakery, and water production. Packaging included production, transport, recycling, internal energy value, and heat recovery. *(MJ per kg bread)*

Note that the small-scale scenarios had somewhat larger energy consumption in processes than their large-scale counterparts. That represents the large-scale effects on energy efficiency that seem to exist (Trägårdh et al., 1979; Nilsson, 1996; Andersson, 1998a).

Packaging can be important as well. The larger energy use for packaging in the large-scale irrational scenario was due to the choice of single-use cardboard boxes for bread distribution, instead of plastic multi--use boxes. The irrational large-scale scenario would have had about equal energy consumption to the irrational small-scale if plastic boxes had been used instead.

The difference between the large-scale scenarios and the small-scale irrational one concerning the transport energy consumption was an effect of scale and distribution area. The large-scale scenarios included a 210-km truck and trailer transport of grain and a 260-km truck transport of bread, which the small-scale did not. The difference between the irrational and rational small-scale systems concerning transport was solely a result of the assumptions made about local bread distribution. In the irrational scenario, it was assumed that the bread was transported 5 km by private car, while in the rational this transport was omitted (done by foot, bicycle or efficient home-distribution system). Like the results of other studies, e.g. Andersson (1998), one can conclude that the assumptions made about the home-transport of food affected the results very much. In this study, that type of transport was more than half of the transport energy consumption in small-scale scenarios and more than a third in the large-scale scenarios. Adding this private car transport to the rational small-scale scenario increased its total energy consumption by almost 30 %, i.e. almost up to the level of the large-scale rational scenario. Thus, it can be concluded that the small-scale system, to have lower energy consumption, must be small-scale also concerning all transport – especially with respect to the last transport from the shop to homes. Another interpretation of the results could be that the scale of the bakery is of less importance concerning energy consumption. It is the efficiency in processes and transport, along with the consumer’s shopping behaviour, that have the largest impact on energy consumption.

Milling of grain typically stands for 2 - 5 % of the total energy consumption. Drying of grain is somewhat more energy consuming, between 4 and 8 % of the total.

Energy consumption hotspots

Energy consumption in all modelled parts is shown in Table 12. All figures are given to two decimal places although that might be to over-estimate the accuracy of the results. In the table, all separate energy consumption figures larger than 10 % of the total are marked as hotspots. Heating of bakery oven and fermentation room, primary packaging and the local distribution and home-transport of bread stood out in most scenarios. Note that if one counts the total contribution of packaging (production, recycling and incineration) it was small in all scenarios except the irrational large-scale one. In that particular case, the choice of cardboard boxes as secondary packaging made the difference.

Table 12. Energy consumption in all parts of the system. Hotspots (>10 % of total) darkened. Zero-values indicate lower consumption than 0.00. Empty places indicate zero consumption, i.e. not used in the scenario. *(MJ per kg bread)*.

	Large-scale irrational (oil)		Large-scale rational (el.)		Small-scale irrational (oil)		Small-scale rational (el.)	
	el.	thermal	el.	thermal	el.	thermal	el.	thermal
Grain Drier	0.02	0.36	0.02	0.29	0.02	0.36	0.02	0.36
Grain Mill	0.26		0.26		0.26		0.11	
Water Production	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Bakery Oven		**2.00**	**1.00**			**2.30**	**1.40**	
Bakery Steam + Ferment.		**1.00**	0.30			**1.50**	0.40	
Bakery Rest	0.80		0.30		0.80		0.20	
Bakery Water Heating	0.21		0.10		0.23		0.23	
Prod. Plastic Bag	0.14	**1.19**	0.14	**1.19**	0.14	**1.19**	0.14	**1.19**
Prod. Plastic Clips	0.00	0.09	0.00	0.09	0.00	0.09	0.00	0.09
Prod. Plastic Box			0.00	0.06	0.00	0.06	0.00	0.06
Prod. Cardboard Box	0.64	**4.22**						
Recycle Plastics[1]	-0.05	-0.61	-0.05	**-0.65**	-0.05	-0.65	-0.05	**-0.65**
Recycle Paper[1]	-0.06	**-2.60**						
Incinerator[2]	0.01	-0.51	0.00	-0.27	0.00	-0.27	0.00	-0.27
"Fill-up" heat[3]				0.23		0.23		0.23
Tractor Transport Grain		0.13		0.13		0.13		0.13
Truck Transport Grain		0.19		0.19				
Regional Distrib. Bread		0.34		0.34				
Local Distrib. Bread		0.94		**0.94**		**0.89**		**0.89**
Priv. Car Transport Bread		**1.23**		**1.23**		**1.23**		
Total [4]	1.98	7.96	2.08	3.76	1.42	7.05	2.47	2.02
Total, el + thermal	9.94		5.84		8.47		4.49	
10 % of the total	0.99		0.58		0.85		0.45	
15 % of the total	1.49		0.88		1.27		0.67	
20 % of the total	1.99		1.17		1.69		0.90	
25 % of the total	2.48		1.46		2.12		1.12	

[1] Negative values imply that energy was saved compared to if incineration had been chosen. Including transport.

[2] Negative values imply that heat was produced. Including transport.

[3] Additional heat production to get the scenarios to produce equal amounts of heat.

[4] Total not exactly the sum of the parts due to rounding-off

Primary energy resources

The three rational scenarios had the lowest consumption of primary energy resources (Figure 64), but the difference between scenarios was smaller than for the energy consumption (Figure 63). It was the choice of electricity as the energy carrier for heating in the baking oven and fermentation room in the rational scenarios that had some impact on the consumption of primary energy, since electricity is more primary energy consuming. If, for example, the rational large-scale scenario were heated by gas the primary energy consumption would decrease from 8.8 MJ/kg bread to just above 7 MJ/kg bread (results not shown). For more details about gas-fuelled rational scenarios, see "Sensitivity analyses".

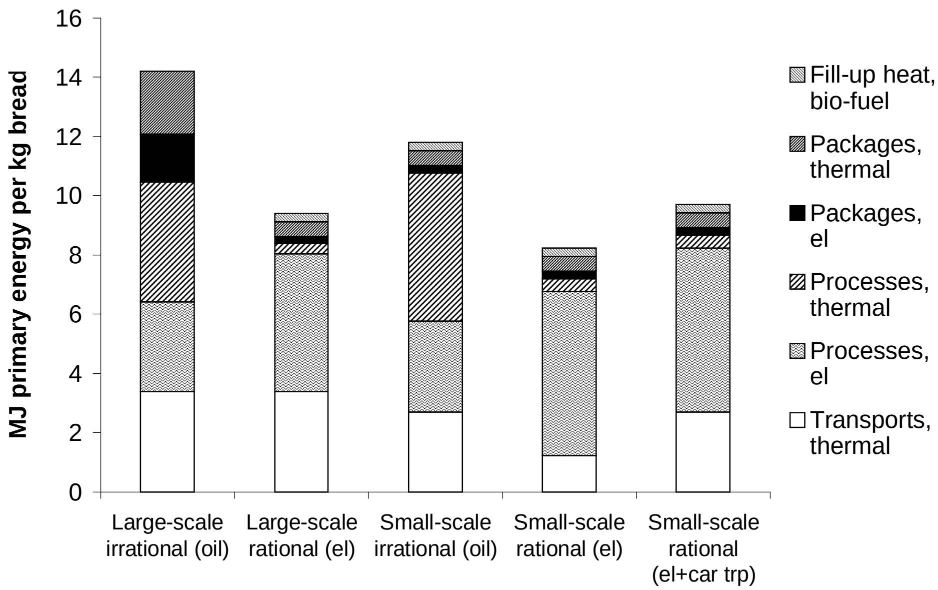

Figure 64. Use of primary energy resources. *(MJ per kg bread)*

Transport did not have any dominant role concerning primary energy use. However, it did make the small-scale rational scenario show better results than the large-scale rational. By inclusion of a 5-km private-car transport in the small-scale rational scenario, it consumed about an equal amount of primary energy resources as the large-scale.

Global warming

The potential global warming impact was largely affected by the choice of energy-carrier for heating of ovens and fermentation-rooms and by use of energy conserving technology. The considerably smaller global warming impact from processes in rational scenarios compared to irrational scenarios (Figure 65) was both due to the use of electricity for the heating of oven and steam boiler, and to rational production and technology. The electricity used was Swedish average, which is mainly hydropower and nuclear power. Choosing e.g. oil-condensed power would reduce the effect of energy carrier choice (see "Sensitivity analyses"). However, energy conserving technology also plays an important role. If the bakery in the rational large-scale scenario had been gas-heated, the process-part would still have shown only a third of the global warming potential compared to the irrational large-scale scenario.

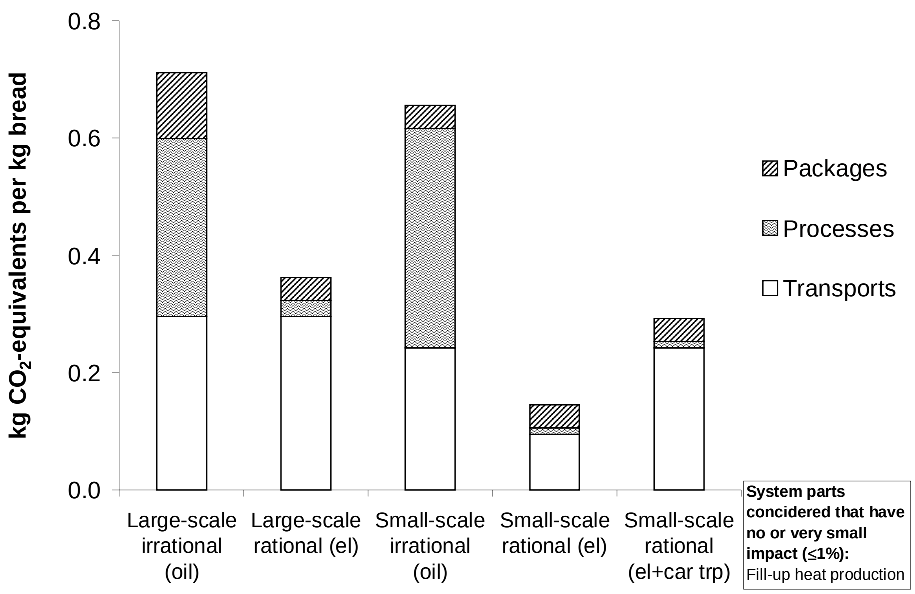

Figure 65. Potential global warming impact (100-years scenario). *(kg CO_2-equivalents per kg bread)*

The small difference between the contribution from processes in both rational scenarios was due to the fact that biofuel was chosen as the energy carrier in the grain-drier in the small-scale system. Furthermore, transport was important concerning the greenhouse effect. In fact, for both rational scenarios transport was the dominating factor. The difference between scenarios concerning global warming was equivalent to those discussed for energy consumption.

Eutrophication

The eutrophication impact was largely owing to emissions of nitrous oxides (from burning of fuels) in this study, i.e. in situations where nitrogen did not have a eutrophication impact, the eutrophication was very small in these systems. Since transport was the dominating NO_x-emitting part, it is not surprising that the small-scale scenarios showed better results for this impact category (Figure 66). For other food-processing systems, e.g. drinking-milk production, which have large direct eutrophying emissions in the water, the result could be different to those obtained here (see "Results and discussion of the liquid milk production system").

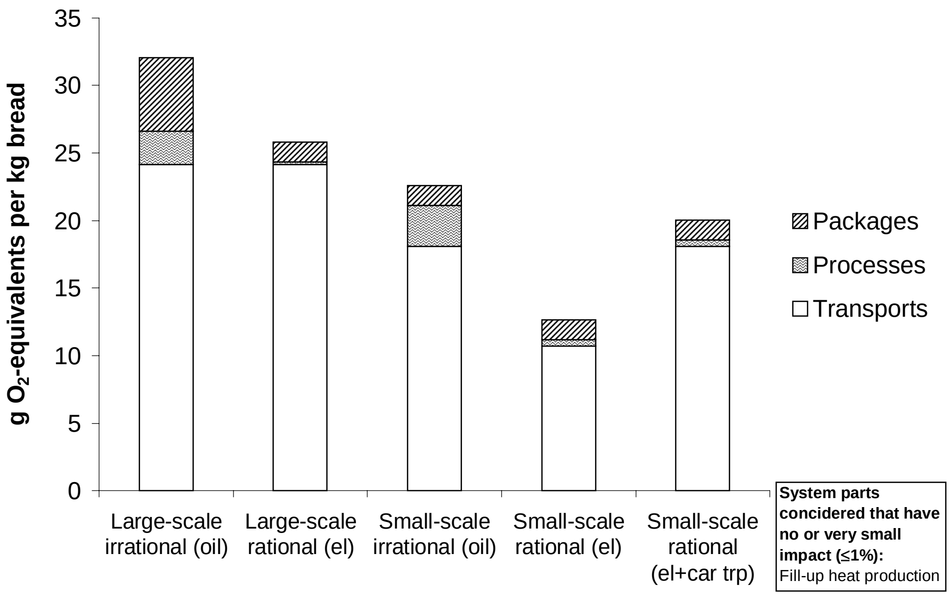

Figure 66. Eutrophication (maximum scenario). *(g O_2-equivalents per kg bread)*

Acidification

In all scenarios, acidification was mainly an effect of emission of nitrous oxides (NO_x) from vehicle engines (Figure 67). In the two scenarios with oil-fired bakery ovens, sulphuric oxides (SO_x) also had considerable impact, i.e. in a situation where nitrous oxides did not have any acidifying impact, the differences between scenarios were larger and the numbers were smaller (Figure 68, please note the difference in scale). The production of cardboard boxes (used for distribution of bread) in the irrational large-scale system contributed to about 10 % of the acidifying effect by emissions of SO_2. However, the cardboard data were from Central-European producers, who use more sulphur-containing fossil fuels in their production than do the Swedish producers. Using Swedish data would decrease the acidifying impact for the irrational large-scale scenario, but it would not change the order of the scenarios.

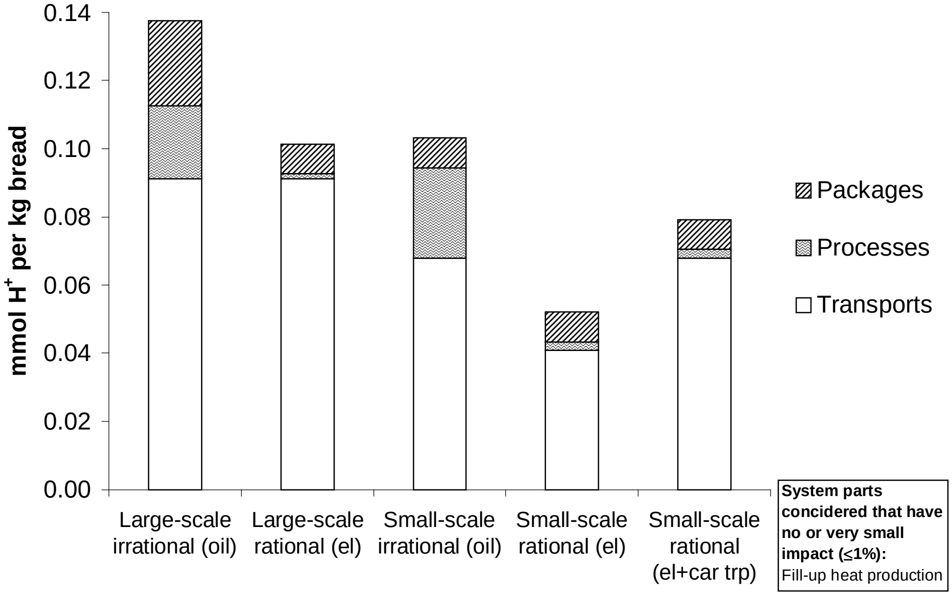

Figure 67. Acidification (maximum scenario). *(mmol H^+ per kg bread)*

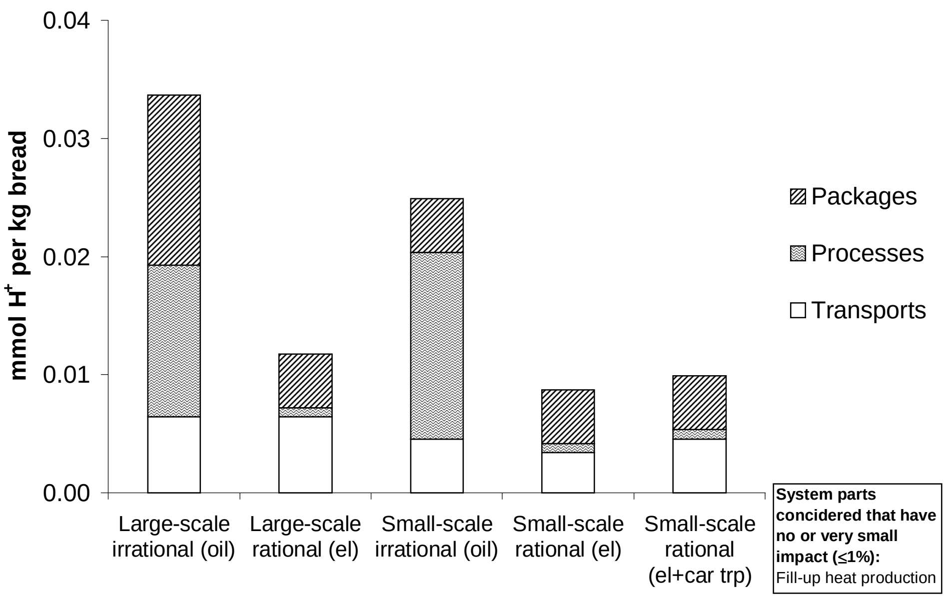

Figure 68. Acidification (minimum scenario). *(mmol H^+ per kg bread)*

Photo-oxidant formation

Concerning photo-oxidant formation too, as can be seen in Figure 69, the rational small-scale scenario had the smallest environmental impact, while there were no large differences between the other three.

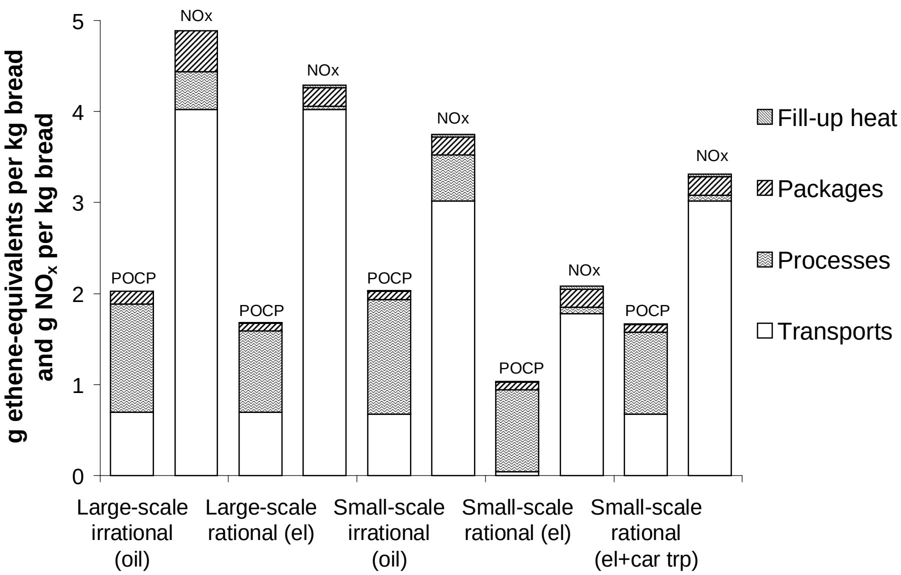

Figure 69. Photo-oxidant formation. *(photochemical ozone creation potentials (POCP) as g ethene-equivalents per kg bread; and g NO_x per kg bread)*

The difference concerning POCP was due to the fact that private cars were not used for home-transport of the bread, i.e. petrol-driven vehicles were the absolutely dominant contributor for POCP looking at the transport alone. It was their approximately ten times larger emission of CO than diesel-fuelled transport which produced that result. The car transport data are valid for city traffic (Eriksson et al., 1995). However, these data are quite old so their accuracy is somewhat questionable. Exhaust catalysts, which are supposed to decrease the emissions of hydrocarbons, are nowadays standard equipment on cars in Sweden. Eriksson et al. (1995) argue that they are aware of that, but that studies from the USA show that the introduction of catalysts does not have as large an effect in city traffic as was expected. The reason is that the exhaust catalysts do not work optimally for the first minutes after the engine is started (5 km shopping trips do not take more than a few minutes), or when the engine has to work hard in e.g. rapid accelerations. They also argue that Swedish conditions, with cold winters, further decrease the performance. In any case, the results concerning the impact of transport on photo-oxidant formation have to be interpreted with the somewhat questionable quality of the data borne in mind.

Release of ethanol in the fermentation process was the dominating contribution from the processes part concerning POCP. In fact, it represented almost 100 % of the processes impact in the rational scenarios, and about 70 % in the irrational. The release of ethanol was set to 3.3 g per kg bread in all scenarios. This is the value used by Andersson (1998) for local bakery and home baking, which is calculated from data given by Pyler (1999). It is of the same order of magnitude as the value given by Kent & Evers (1994), and thus was chosen since better data were not available. The ethanol release was equal in all scenarios. Therefore, the difference between the two large-scale systems could be concluded to depend on the choice of fuel (oil and electricity respectively) for oven heating and steaming in the bakery. Since the small-scale irrational scenario had larger energy consumption in bakery processes, and as its shorter diesel-fuelled transport did not result in a smaller impact, it had an equal POCP impact to the irrational large-scale scenario. The NO_x-emissions, as discussed under "Acidification", largely originated from the combustion of fuel in vehicle engines, so transport was totally dominant for these emissions.

Summary of results, bread production system

All impact categories reported are listed for all scenarios in Table 13. As can be seen, the small-scale rational scenario showed the best results for all impacts, except for water consumption. It might be possible to decrease this consumption, but I chose to keep it high in both small-scale scenarios. Anyway, the environmental effect of this would be small.

Table 13. Overall results for different bread production scenarios. *(impact per kg bread)*

	Large-scale irrational	Large-scale rational	Small-scale irrational	Small-scale rational	Small-scale rational + car transport
Global warming, GWPs *(kg CO_2-equivalents)*	0.71	0.36	0.66	0.15	0.29
Acidification, max *(mmol H^+)*	0.138	0.102	0.104	0.053	0.080
Acidification, min *(mmol H^+)*	0.034	0.012	0.025	0.009	0.010
Photo-oxidants, POCPs *(g ethene-equivalents)*	2.0	1.7	2.0	1.0	1.7
Photo-oxidants, NO_x *(g NO_x)*	4.9	4.3	3.7	2.1	3.3
Eutrophication (g O_2-equivalents)	32	26	23	13	20
Energy consumption *(MJ electricity)*	2.0	2.1	1.4	2.5	2.5
Energy consumption *(MJ thermal)*	8.0	3.8	7.1	2.0	3.3
Primary energy cons. (MJ primary energy resources)	14	9	12	8	10
Water consumption *(litre water)*	1.4	0.8	1.5	1.5	1.5

The results are also shown relative to the large-scale irrational scenario in Figure 70. A large part of the difference between the large-scale and small-scale rational scenarios can be explained by the choice of transport from retailer to home. However, the small-scale rational scenario still showed better results for global warming, acidification, NO_x-emission and eutrophication compared to the large-scale rational scenario even if the home-transport was carried out by car. Thus, it can be concluded that the rational small-scale system had a smaller environmental impact but about equal energy consumption compared to the rational large-scale system. This indicates that emissions from transport are large.

However, the emission data are a few years old and could be a source of error. This is an area that should be followed up more closely.

It can also be seen that the irrational small-scale scenario had a larger environmental impact for all categories compared to the rational large-scale scenario. Thus, it can be concluded that a "bad" small-scale system is worse concerning these environmental impacts than a "good" large-scale system.

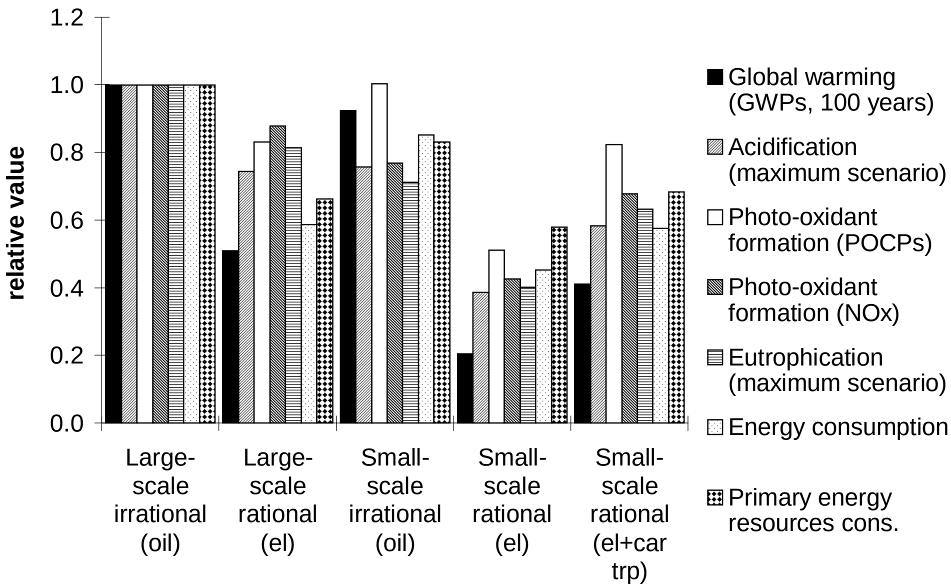

Figure 70. All impact categories relative to the impact of the large-scale irrational scenario. *(relative values, 1=Large-scale irrational scenario)*

Sensitivity analyses; bread production system

Fill-up heat from fossil oil

The environmental impacts included in this study were not affected on a system level when the fill-up heat was produced from oil instead of from biofuels (results not shown).

Electricity produced in oil-fired power plant

Since Swedish average electricity is not comparable to electricity produced in most countries in the world, it might be of interest to run the scenarios with electricity produced in oil-fired power plants. This is a type of electricity production used in Sweden for marginal supply. It is also a "dirtier" type of production, concerning the environmental impacts included here.

Global warming

Choosing oil-condensed power reduced the effect of the different energy carriers used for oven heating concerning global warming impact (Figure 71 compared to Figure 65, please note the difference in scale). The order of the four main scenarios was not changed, but the fifth scenario (rational small-scale + car transport) went from a somewhat smaller impact to about equal impact compared to the rational large-scale scenario. The total global warming impact was significantly increased in all scenarios.

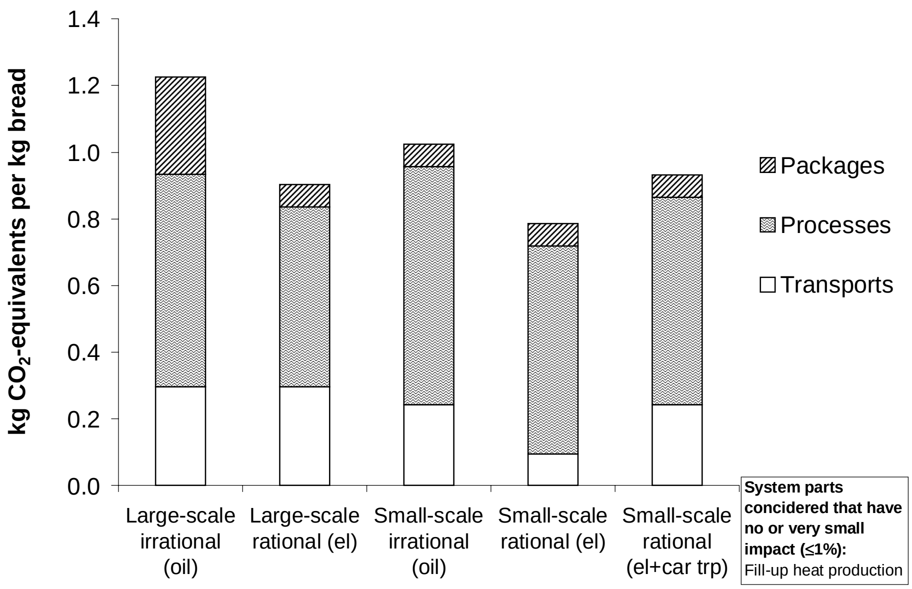

Figure 71. Potential global warming impact (100-years scenario). Electricity produced in oil-fired power plant. *(kg CO_2-equivalents per kg bread)*

Acidification

For acidification, the choice of oil-fired electricity production had large implications (Figure 72 and Figure 73, compared to Figure 67 and Figure 68, please note the differences in scales).

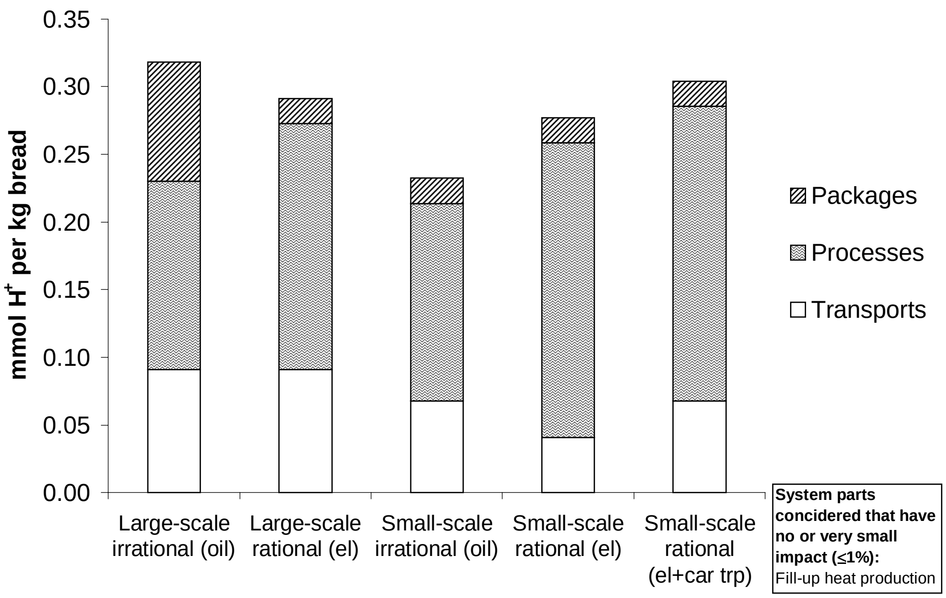

Figure 72. Acidification (maximum scenario). Electricity produced in oil-fired power plant. *(mmol H^+ per kg bread)*

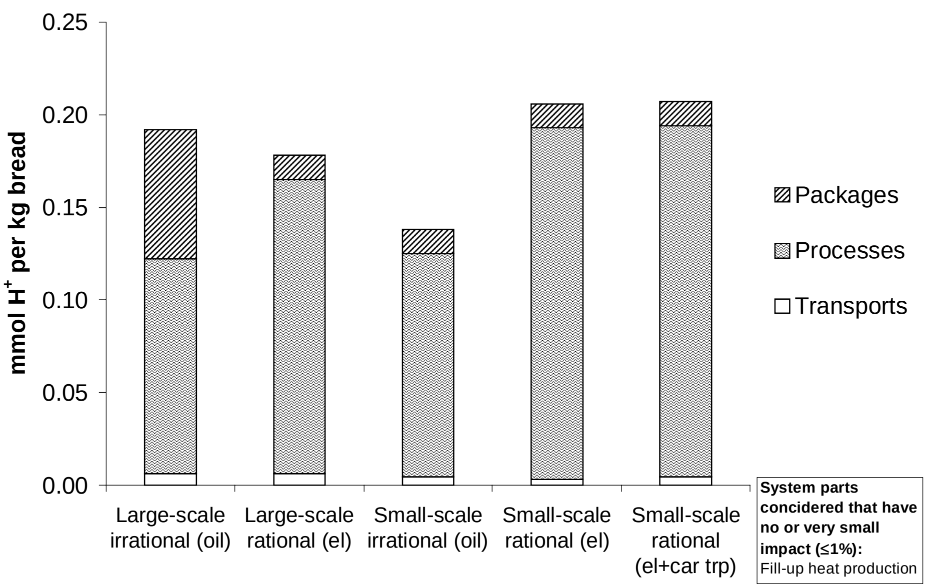

Figure 73. Acidification (minimum scenario). Electricity produced in oil-fired power plant. *(mmol H^+ per kg bread)*

The total acidifying impact was significantly increased in all scenarios. The order between the scenarios was changed but the differences between them were levelled out. With this kind of electricity, the irrational small-scale scenario showed the best results, while the rational small-scale scenarios had an acidifying impact of the same magnitude as the large-scale scenarios.

For the maximum acidifying scenario (where nitrogen compounds contributed) the increase in impact was 2 – 5-fold, while for the minimum acidifying scenario the changes were between 6 and 20-fold. This was a result of sulphuric emissions in the combustion of oil. Using oil with low-sulphur content reduced this effect.

Photo-oxidant formation

For photo-oxidant formation impact, the choice of oil-fired power production had lower effects (Figure 74, compare to Figure 69). The impact was increased in all scenarios but the order of the scenarios was not changed.

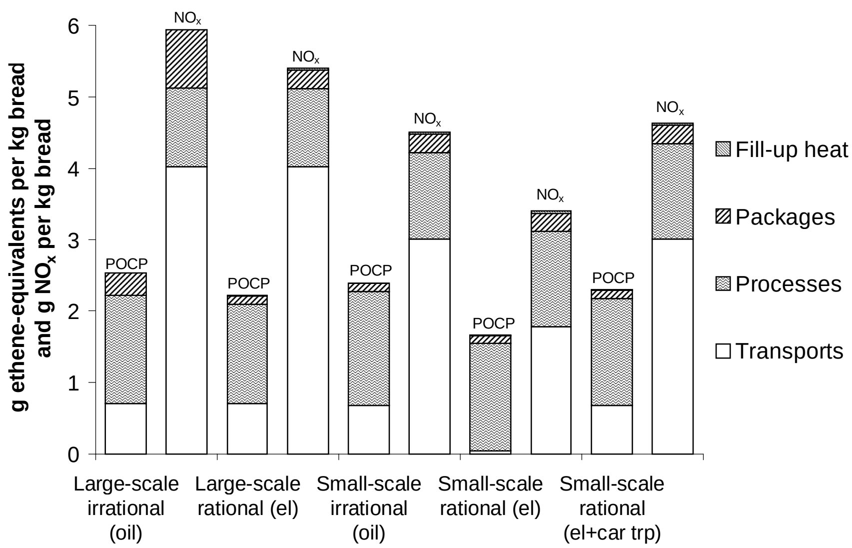

Figure 74. Photo-oxidant formation. Electricity produced in oil-fired power plant. *(photochemical ozone creation potentials (POCP) as g ethene-equivalents per kg bread; and g NO_x per kg bread)*

Eutrophication

For eutrophication, the choice of oil-fired power production had quite a small effect (Figure 75, compare to Figure 66). The impact was increased in all scenarios but the order of the four main scenarios was not changed. However, the eutrophication impact for the rational small-scale scenarios increased more compared to the irrational small-scale scenario, resulting in a slightly larger eutrophication impact for the fifth scenario compared to the irrational counterpart.

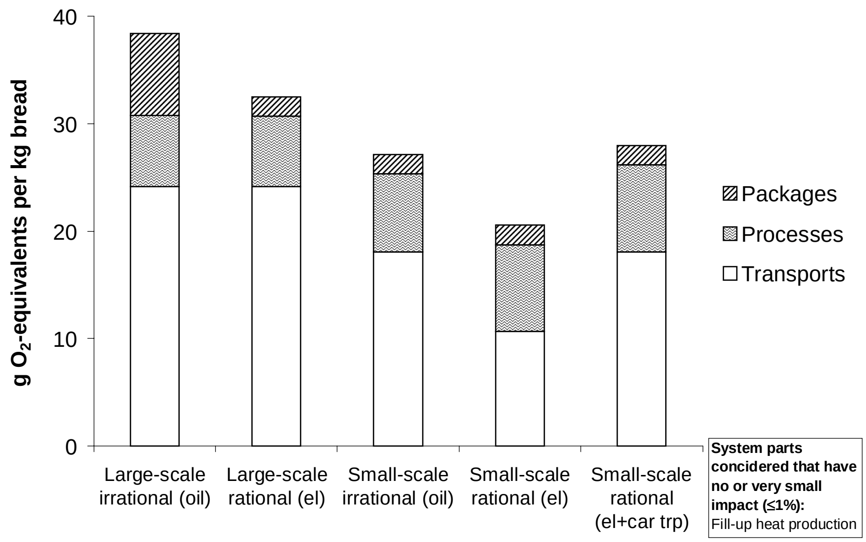

Figure 75. Eutrophication (maximum scenario). Electricity produced in oil-fired power plant. *(g O_2-equivalents per kg bread)*

Summary, sensitivity analysis of oil-fired power production

Figure 76 shows that the differences in environmental impacts were smaller when electricity was produced in oil-fired power plants compared to being produced as Swedish average (Figure 70). The trend was still, however, that the rational small-scale scenario showed the smallest impacts. However, adding the private car transport made the rational small-scale scenario level with the rational large-scale scenario.

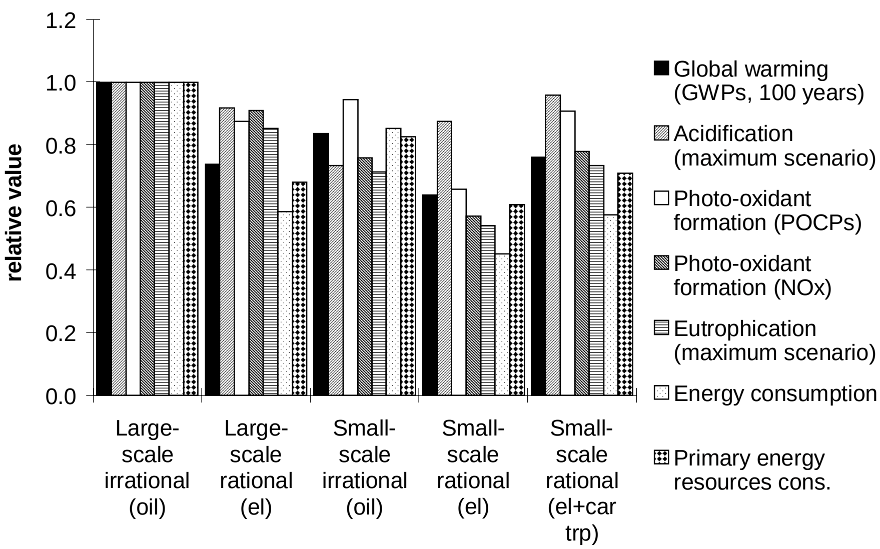

Figure 76. All impact categories relative to the impact of the large-scale irrational scenario. Electricity produced in oil-fired power plant. *(relative values, 1=Large-scale irrational scenario)*

Rational scenarios gas-fired instead of electrically heated

In Sweden, where electricity is mainly produced from hydropower and nuclear power, it seems rational to use electricity for many purposes. However, for many other places this is not that self-evident. The rational bread production scenarios in this study were, therefore, run in a set-up where the heating of ovens and fermentation rooms was done by burning of fossil gas. In the small-scale rational scenarios, the grain drying process was also modified from an indirect biofuel-heated to a direct gas-heated drier. The irrational scenarios were not changed. By comparing Figure 77 and Figure 70, one can conclude that the shift from electricity to fossil gas did not produce any dramatic changes in environmental impacts. Only the global warming was affected, in the large-scale scenario by about 25 %. For the small-scale scenario, the increase was almost 100 %, partly due to the changes in the grain-drying set-up. In the small-scale scenario where private cars were used for home-transport, the increase in global warming impact was about 50 %.

The other effect that could be observed was on the consumption of primary energy resources. Here, the fossil gas set-up was better. The consumption decreased by almost 20 % in the large-scale scenario, by 25 % in the (first) small-scale scenario, and by about 20 % in the small-scale scenario where private cars were used for home-transport. This is an indication that use of electricity for heating purposes is not an efficient use of resources. However, depending on how the electricity was produced and on which resources were scarce and which were abundant, electricity nevertheless fell out as the best alternative.

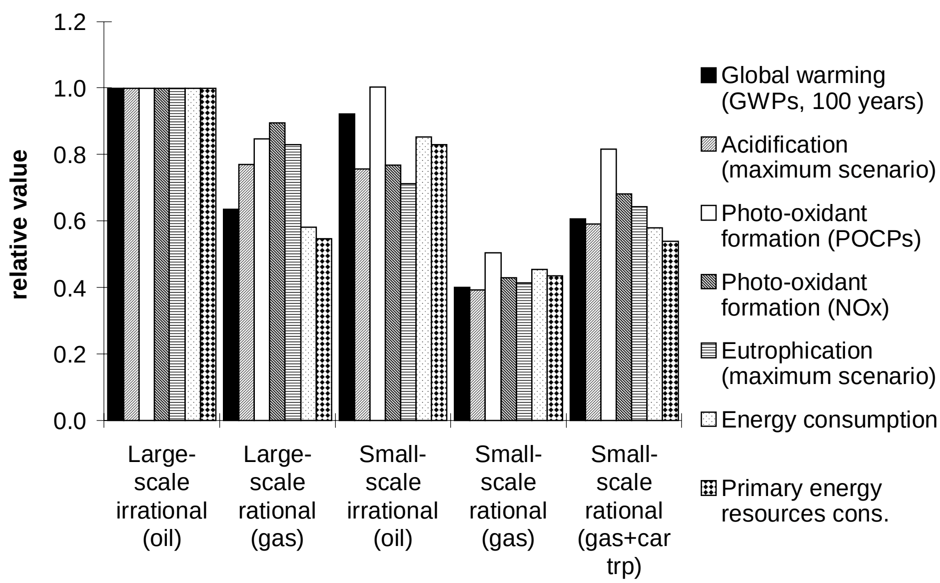

Figure 77. All impact categories relative to the impact of the large-scale irrational scenario. Rational scenarios gas-fired. *(relative values, 1=Large-scale irrational scenario)*

Imported grain

Imported grain is sometimes used to get a larger protein (gluten) content in the flour, but there seems to be no difference between scales. Possibly it is more an indicator of how well the bakery process is designed, since a gently-handled dough requires flour with lower protein content to make a fluffy bread than does a roughly-handled dough. To say that it is correlated to scale or energy efficiency is probably not relevant. Therefore, no sensitivity analysis with use of imported grain was performed.

Liquid milk production system

This chapter presents an introductory study of the liquid milk production system. The aim was, as mentioned before, to study a common type of food that could be expected to have different properties concerning energy consumption and environmental impacts compared to bread production. The presentation starts with methodological issues and continues with the results and discussion.

System boundaries

This study followed the structure of the bread production study. It included transport of milk from farm to dairy, processing of milk (dairy total energy consumption), clean water production, production of packaging for milk, handling of waste milk packaging, sewage plant treatment of dairy wastewater, and transport of liquid milk from the shop to the consumer's door. The system, as studied here, is shown in Figure 78.

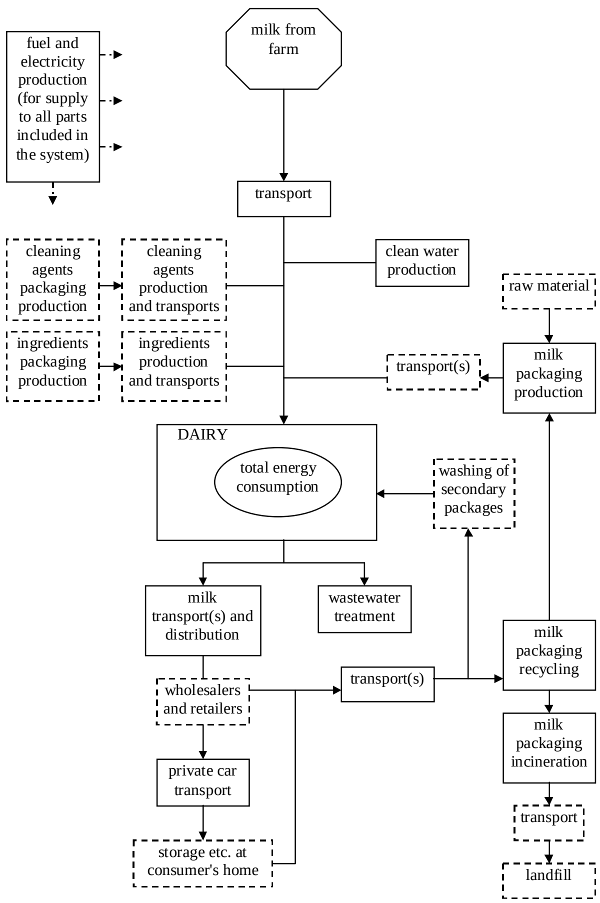

Figure 78. Flow chart of the milk production system. Dotted blocks not included in the study.

In order to give results comparable to the results from the bread production study, the system boundaries were as equal to those in the bread production study as possible. Thus, the primary production of milk was not included in the analysis, neither were the retailer level nor the consumer phase, although they are probably of more importance in the milk system than in the bread system. Other parts of the system left out were on-farm storage of milk and the fate of dairy by-products. Concerning packaging, the transportation of packaging from manufacturer to dairy was not included. Since same package type was chosen in all scenarios, no "fill-up" heat production was needed.

The model outputs were yearly amounts of emissions and energy consumption, but all results were recalculated to emissions and energy consumption per kg milk produced. To obtain the results for a 100-person unit, the per-kg-milk results have to be multiplied by 15 000; 150 kg milk per person and year in Sweden (Becker, 1992) times 100 persons.

Scenarios

As in the bread production study, four main scenarios were presented. Two were large-scale systems and two small-scale systems. Additionally, a fifth scenario was included, namely the small-scale rational scenario complemented with the private-car transport used in the other scenarios. The processed milk was distributed in cardboard single-use packages in all scenarios. The cardboard was assumed to be recycled, eventually ending up in an incineration plant. Dairy wastewater was treated in a sewage plant in all scenarios.

1. *Irrational large-scale*, was not energy-optimised and the dairy used oil for heat production. Milk was transported 100 km by truck to the dairy. The processed milk was first transported 100 km by truck (inter-regional), then locally distributed to shops by truck (100 km round-trip), and finally transported 5 km by private petrol-driven car to the consumer's home.

2. *Rational large-scale*, had an energy-optimised dairy. It used electricity for all heating. Transport was equal to that in the irrational large-scale scenario.

3. *Irrational small-scale*, had a non-energy-optimised dairy that used oil for heat production. The milk was transported 50 km by truck to the dairy. The processed milk was distributed locally to the shops by truck (100 km round-trip) and then, as in the large-scale scenarios, transported to the consumer's home by private car.

4. *Rational small-scale*, used electricity for all heating in the dairy, which was energy optimised. The transport to the dairy and the distribution to shops were equal to those in the irrational small-scale scenario. The transport from the shop to the consumer's home was done on foot, by bicycle, or by home-distribution carried out by the dairy. In the latter case this was assumed to be included in the energy consumption calculated for the local distribution, and thus assumed to be carried out in an energy efficient manner.

5. *Rational small-scale + car transport*, was equal to the fourth scenario but was complemented with the vehicle-transport to the consumer's home by private (petrol-driven) car as used in the first three scenarios.

Models

The general structure of the model was equivalent to that in the bread production study. Where appropriate, the same sub-models were used. The dairy model is structured like the bakery model, but no partition of energy consumption between different processes was performed, i.e. total energy consumption (including heating of premises etc.) for dairies was used. The model makes it possible to choose electricity and/or fuel as the energy source. For air emission calculation, it is possible to choose different fuels for the heat production. The water emissions are calculated in the model. Concerning the transport from the shop to the consumer's home by private car, the same assumptions were made as in the bread production study. Liquid milk, like bread, stands for about 6 % of the cost of our food consumption (SCB, 1999), and thus 6 % of the transport was allocated to the liquid milk. Data for energy consumption, transport distances, water consumption and water emissions are collected from several sources (Elsy, 1981; Cox & Miller, 1985; Lucas, 1985; Verma, 1988; Larsson, 1997; Høgaas Eide, 1998; Hedegård, 1999; Bengtsson, 1999; Pyler, 1999; Jürss, 1999). For more details see Appendix 8.

Results and discussion of the liquid milk production system

For liquid milk production systems, the small-scale scenarios had no advantages concerning energy consumption (Figure 79). The lower energy consumption for transportation did not compensate for energy consumption in processes, as was the case in the bread production system (Figure 63). However, with regard to the environmental impacts, the situation was different. As an example, the global warming impact is shown in Figure 80. Transport was a more dominant part for environmental impacts than for energy consumption. The same was true for acidification, eutrophication and emissions of NO_x. Whether this was due to real differences or weaknesses in data reliability might be a question for further research.

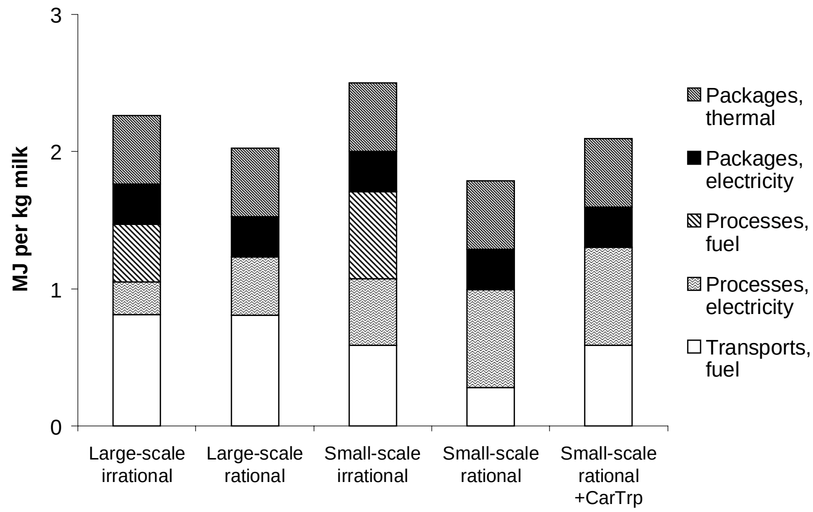

Figure 79. Energy consumption in different system parts. *(MJ per kg milk)*

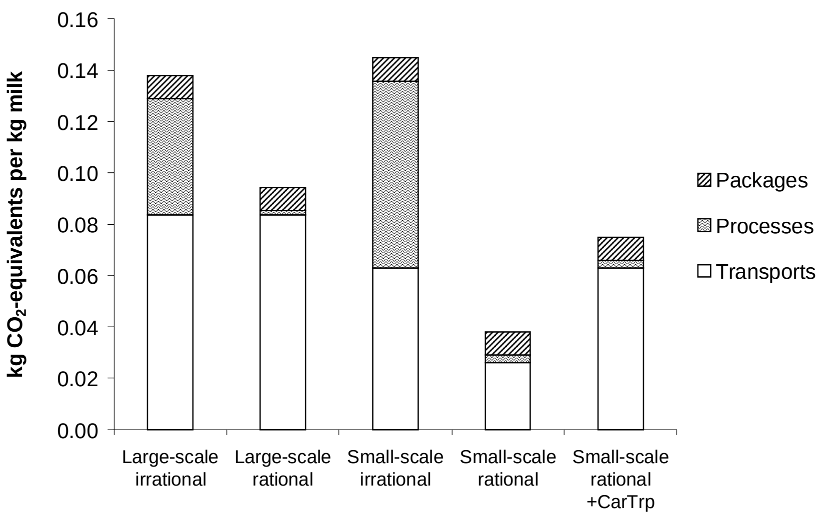

Figure 80. Potential global warming impact (100-years scenario) due to emissions from different parts in the system. *(kg CO_2-equivalents per kg milk)*

The environmental impact categories assessed are listed for the five scenarios in Table 14. The results are also shown relative to the large-scale irrational scenario in Figure 81. It could be concluded that the tendency was similar to that in the bread production study. Small-scale systems can be better concerning the impact categories included, *but* to make significant improvements the small-scale has to be combined with a local market, i.e. the consumers must be able to walk or bike to the shop, or have the food (energy-efficiently) delivered at the doorstep.

Table 14. Overall results for the different liquid milk proLduction scenarios. *(impact per kg milk)*

	Large-scale irrational	Large-scale rational	Small-scale irrational	Small-scale rational	Small-scale rational + car transport
Global warming, GWPs *(kg CO_2-equivalents)*	0.14	0.09	0.15	0.04	0.08
Acidification, max *(mmol H^+)*	0.033	0.030	0.026	0.014	0.021
Acidification, min *(mmol H^+)*	0.005	0.004	0.006	0.003	0.003
Photo-oxidants, POCPs *(g ethene-equivalents)*	1.1	1.1	1.1	0.9	1.1
Photo-oxidants, NO_x *(g NO_x)*	1.3	1.3	1.0	0.6	0.9
Eutrophication (g O_2-equivalents)	9.6	8.7	8.5	4.1	5.9
Energy consumption *(MJ electricity)*	0.5	0.7	0.8	1.0	1.0
Energy consumption *(MJ thermal)*	1.7	1.3	1.7	0.8	1.1
Primary energy cons. (MJ primary energy resources)	3.3	3.2	3.9	3.3	3.7
Water consumption *(litre water)*	2.0	1.0	4.0	3.0	3.0

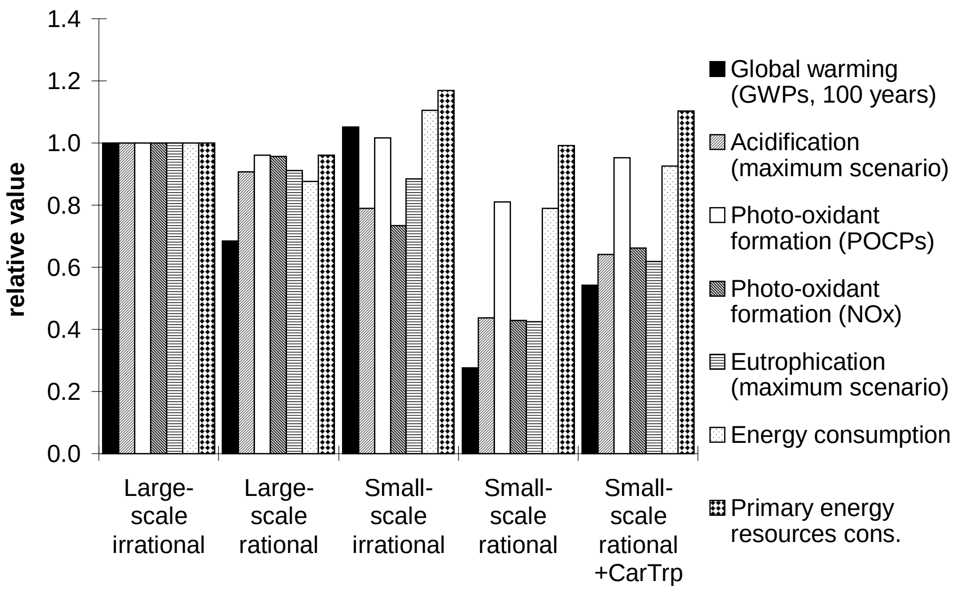

Figure 81. All impact categories relative to the impact of the large-scale irrational scenario. *(relative values, 1=Large-scale irrational scenario)*

Sensitivity analyses; milk production system

Electricity produced in oil-fired power plant

A sensitivity analysis where oil-fired power plant electricity was used is shown in Figure 82. It can be noticed that the rational scenarios increased their global warming and acidifying impacts compared to when electricity was produced as Swedish average. Other impacts were affected to a lower extent.

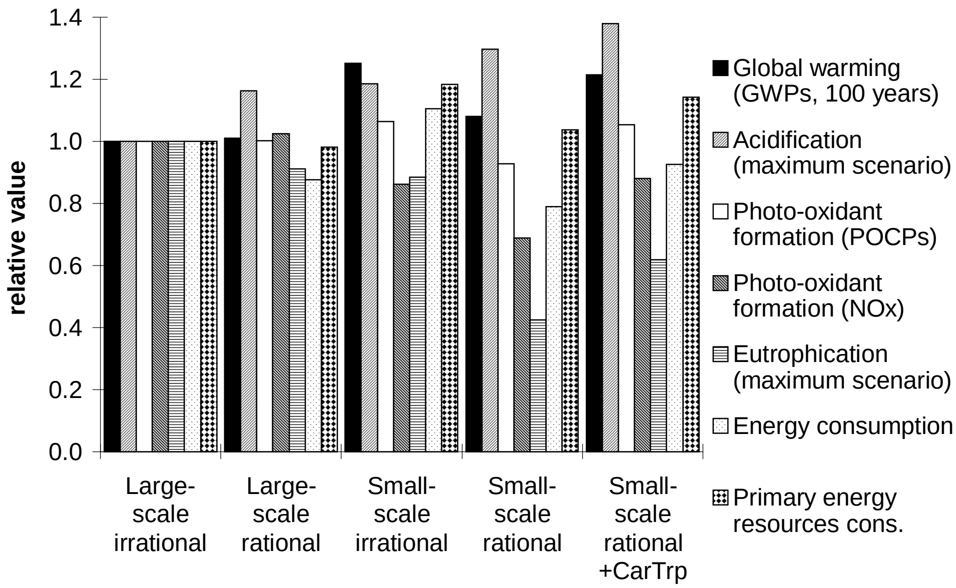

Figure 82. All impact categories relative to the impact of the large-scale irrational scenario. Electricity produced in oil-fired power plant. *(relative values, 1=Large-scale irrational scenario)*

Processed milk not transported inter-regionally

To assume that the processed milk was not transported inter-regionally (i.e. 100 km shorter truck transport) in the large-scale scenarios had some impact on the results. The total thermal energy consumption decreased by about 10 %, resulting in decreased primary energy consumption by 5 %. Global warming, acidification, NO_x-emissions, and eutrophication decreased by 10-20 %.

Cyclic small-scale food systems – an example

Though far from complete, I will present an example of a cyclic small-scale food system and compare it to a current system. The aim is to show the maximum effect that could be obtained if a chain in a food-system were changed to a cyclic small-scale system. The parts considered in the system were organic waste management, N emissions from soil, grain-production field operations, chemical fertiliser production, grain and bread processing, clean water production, production of packaging and transport between all parts in the system, i.e. all parts considered in the previously presented chapters “Organic waste management system” and “Bread production system”. The example reflects one chain of the flow of organic matter that goes from-consumer-via-agriculture-to-consumer.

The perspective when putting together the example, called the Cyclic-Small-Scale-scenario, was to choose a system that was thought to be resource conserving and energy efficient. It was compared to the Situation-of-Today-scenario, which was thought to represent the large-scale and nonenergy-optimised system that exists today, though many systems are evolving to become large-scale *and* efficient.

The results for the two scenarios were obtained by summation of results from scenarios in the studies of organic waste management and bread production presented earlier in this thesis. The Situation-of-Today-scenario was the sum of the “SludgeToSewPlant” organic waste management scenario and the “Large-scale irrational” bread production scenario. The Cyclic-Small-Scale-scenario was the sum of the “AnaerobicDigestionUrineSeparationNoHygienisation” organic waste management scenario and the “Small-scale rational” bread production scenario.

A small-scale scenario containing anaerobic digestion could be argued to be very hypothetical but small-scale digestion is getting increased attention in e.g. Austria, Germany, Denmark, and Sweden (Lindberg & Edström, 1998). The results for the Cyclic-Small-Scale example are presented in Figure 83 as relative values compared to the Situation-of-Today scenario. For actual contribution to impact categories see Table 15.

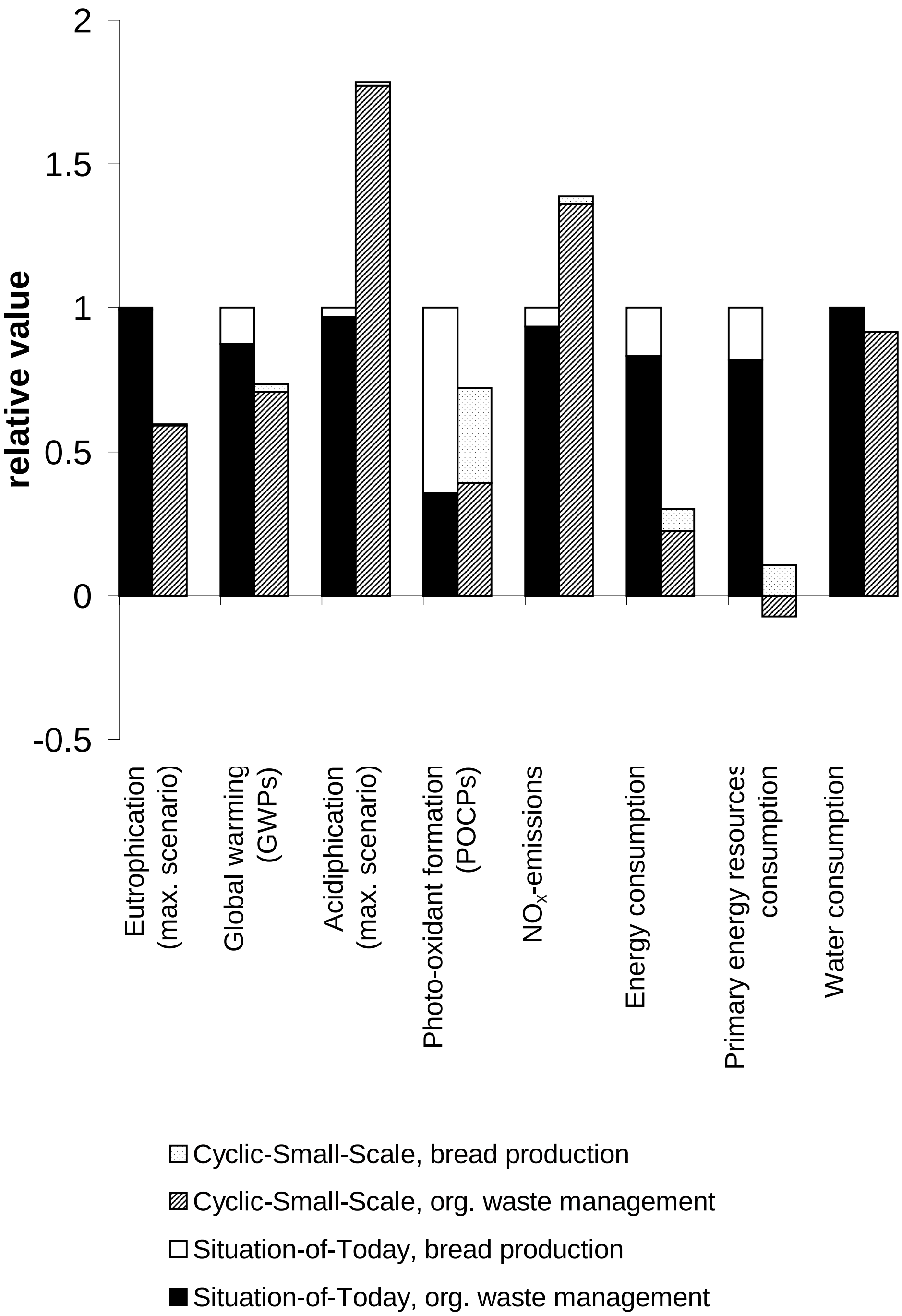

Figure 83. Environmental impacts and resources consumption. Cyclic-Small-Scale scenario compared to Situation-of-Today scenario. *(relative value, 1=Situation-of-Today)*

Table 15. Environmental impacts and resources consumption. Organic waste management system and bread producing system serving 100 people. *(total result per 100 people and year)*

	Situation-of-Today-scenario			Cyclic-Small-Scale-scenario		
	Organic waste manage-ment *SludgeTo SewPlant*	Bread produc-tion *Large-scale irrational (oil)*	Total [1]	Org. waste manage-ment *AnaerobDi gUrineSep NoHygien-isation*	Bread produc-tion *Small-scale rational (el)*	Total [1]
Global warming, GWPs *(kg CO_2-equivalents)*	17 400	2 500	20 000	14 100	520	15 000
Acidification, max *(mol H^+)*	14.8	0.5	15	27.1	0.2	27
Acidification, min *(mol H^+)*	0.9	0.1	1.0	3.3	0.0	3.4
Photo-oxidants, POCPs (kg ethene-equivalents)	4.0	7.2	11	4.3	3.7	8.0
Photo-oxidants, NO_x *(kg NO_x)*	250	20	270	360	10	370
Eutrophication (kg O_2-equivalents)	21 700	100	22 000	12 900	50	13 000
Energy consumption *(MJ)*	175 000	35 000	210 000	47 000	16 000	63 000
Primary energy cons. (MJ primary energy resources)	227 000	50 000	280 000	-20 000	29 000	9 000
Water consumption *(m^3 water)*	7 300	5	7 300	6 700	5	6 700

[1] Rounded off to two valid figures

Although the Cyclic-Small-Scale scenario showed much better results for eutrophication, global warming, and photochemical ozone creation potential impacts, it cannot be said to be the ultimately best alternative concerning environmental impacts, since its acidifying impact was almost doubled. In addition, the emission of nitrous oxides increased by about a third. These effects were mainly due to changes in organic waste management.

Regarding energy and primary energy resources, the Cyclic-Small-Scale system chosen represented substantially lower net consumption due to the production of biogas. The negative value for primary energy consumption (i.e. primary energy production) in the organic waste management can be explained by the fact that the gas was partly used for electricity production. This was assumed to substitute for electricity produced as Swedish average electricity, which would have consumed 2.35 MJ primary energy resources for each MJ produced. The equivalent value for electricity produced in an oil-fired power plant is 2.69 MJ/MJ.

Concerning acidification, it is important to remember that in many ecosystems in Scandinavia and Northern Europe, nitrogen compounds are usually neutralised to a large extent. Therefore, figures for an acidification minimum-impact scenario are also presented in Table 15. These indicate that the acidification could be substantially smaller than the maximum acidification scenario. In practice, the situation should be somewhere in between the two. In the minimum-acidification case, the Cyclic-Small-Scale system would have an approximately three times larger acidifying impact compared to the Situation-of-Today system. This is due to the fact that the acidification was largely an effect of nitrogen compounds (not counted in the minimum acidification scenario) in the Situation-of-Today system, while in the Cyclic-Small-Scale system it was sulphuric oxides from the combustion of biogas which caused the effect. As mentioned above, this can be dealt with by process adjustments that decrease the creation of hydrogen sulphide or by cleaning of the biogas before combustion.

The emissions of NO_x largely emanated from transport.

General discussion

This thesis is an attempt to evaluate some environmental impacts and consumption of some natural resources in small-scale cyclic food-systems. Although the entire food-system has not been included, some general conclusions can be drawn. The most self-evident conclusion is, as expected, that no single system or scenario is “best” concerning all impact categories. There is always a drawback, which has to be taken into consideration in the process of a real-world choice of system. For example, the Cyclic-Small-Scale scenario presented in previous chapter showed substantially better results for energy turnover and for most of the environmental impacts studied, but had double the contribution to acidification compared to the Situation-of-Today scenario and also somewhat greater NO_x-emissions. Thus if acidification is an environmental problem in the area/region of concern, maybe this system should not be chosen or, at least, actions to avoid the problem will have to be taken into account. The on-site impacts of NO_x have also to be examined. Furthermore, it might be proper to make a normalisation against total emissions from other sources in the area/region to assess the relative contribution of the system under investigation in its context.

When making a study for later implementation, it is essential to choose scenarios for impact assessment, types of fuels and electricity production types that are relevant for the area and situation of concern, as these choices may actually determine the results. Moreover, it is important to be clear about the purpose and goal of the study and about the timeframe one has in mind for possible system modifications.

Organic waste management

Concerning organic waste handling, the scenarios studied represented systems present or possible in small towns, villages and rural areas, i.e. no scenario with sewage plant treatment of wastewater was included. I would still claim that many of the results are general in their character. Firstly, one can conclude that, in general, the household residue part is small compared to the internal flows in agriculture. However, it might be important to recycle these residues in regions with small numbers of animals, or with large numbers of people. Furthermore, the recycling of municipal organic waste can serve as a means to get better treatment for the on-farm organic residues (e.g. manure) as it may change the attitude to organic fertilisers and give opportunities for co-treatment of agricultural and municipal organic wastes. It could also be an economic sideline for farmers to take care of and treat municipal organic wastes.

When comparing different sewage systems, three main criteria are of concern – sanitary performance, conservation of environmental and natural resources assets and technical and socio-economic aspects (Finnson et al., 1995). Included in technical and socio-economical aspects are e.g. user friendliness, impact on social planning, questions of responsibility, and economics. Although environmental impacts are focused upon here, the other aspects have to be remembered when real-life case studies are performed.

As expected, plant nutrient recycling is improved when reactor treatments of blackwater and organic wastes are used, compared to on-site sludge separation and biopond treatment of unsorted household wastewater in combination with home composting of organic wastes. However, the study shows that it is important to make an analysis of the plant-availability properties of the material brought back to arable land, and not only of its amounts, when studying plant nutrient recycling. This may be especially valid for the different fractions of nitrogen, but phosphorus also seems to have different qualities (see e.g. Hagström et al., 1997; Linderholm, 1997; Ugland et al., 1998). By calculating expected yield of grain resulting from the amounts of different fractions of nitrogen recycled, which have different fertilising efficiencies, I have shown large differences in performance between treatments that give comparable results when looking at amounts recycled. Thus, when analysing plant nutrient recycling it is important to remember that the social residues are going to be utilised in agriculture. Instead of calculating the expected yield of grain, one might recalculate the recycled nutrients to chemical fertiliser equivalents but then it is easy to lose one's perspective on conservation and self-sufficiency with respect to locally-available resources. However, knowledge about the long-term effects of organic waste recycling for the soil is limited. More research is needed to be able to assess the quality of organic fertilisers originating from social wastes. Furthermore, the plant nutrients that are not utilised after recycling might turn out to be pollutants in surrounding ecosystems.

Somewhat surprisingly, the liquid composting treatment resulted in significantly smaller yields of grain than the anaerobic digestion treatment. This was probably a result of imperfections in the model, but more research is needed to evaluate the reason.

Urine-separating toilet systems increased both the recycling of nutrients to arable land and the expected yield of grain. The effect was most obvious in the on-site sludge separation scenarios where it gave dramatic improvements (seven to eight times larger yield of grain). For the biopond and compost treatments, urine separation gave about three times larger additional yield of grain while in the liquid compost and anaerobic digestion scenarios it had a minor effect. The introduction of urine-separating toilets into the systems studied resulted in equal or smaller energy consumption. The water saving achieved in urine-separating toilets resulted in decreased fuel consumption for transportation and spreading of the household residues by about a quarter in liquid compost and anaerobic digestion scenarios. However, the largest energy saving effect in real terms was obtained internally in those treatments. Other water-saving technologies would probably give a similar result.

The liquid compost and anaerobic digestion scenarios had large energy consumption for transportation compared to the others, due to the large volume (mostly water) handled. In spite of this, these scenarios showed better results when it came to net energy turnover for the entire system, due to the energy production obtained in the treatments. In places where pipe-transport is possible, the energy consumption may be further decreased. Dry composts produce heat that should also be possible to utilise, for example by condensation of outlet gases. This option was, however, not included in this study. By proper treatment of the condensation water, it should be possible to use it as nitrogen fertiliser.

The anaerobic digestion scenario with urine-separating toilet system and no hygienisation of incoming material showed figures which indicated that a net energy break-even was possible to reach. Note that this is valid with the 2-km transporting distances chosen in this study. Still, some energy saving in the system assumed here would have to be performed to reach zero-energy-consumption. Choosing 10 km average transporting distances increased energy consumption by about 30 % in the liquid compost scenarios, and by between 35 and 55 % in the anaerobic digestion scenarios (energy production not affected). For the other scenarios, the change was in the order of 10 %. It might not be economically viable to use small-scale anaerobic digestion treatment today because of the high investment cost, but low-tech small-scale digestion is getting increased attention in several countries (Lindberg & Edström, 1998). By co-treatment of social and agricultural wastes, digestion seems likely to become an alternative for farmers to be less dependent on fossil fuels.

It is at the same time worth noticing that the anaerobic digestion scenario combined with an ordinary sewage system and hygienisation of incoming material had the most negative energy turnover of all scenarios. On the positive side for biogas production is the fact that the digestion reactor requires heat (and some electricity for pumping) but produces gas, which has "higher energy quality". This gas can for example be used for electricity (and heat) production or cleaned and compressed for use as vehicle fuel. The equivalent drawback for liquid composting is that it demands electricity but produces heat. Besides, it is not easy to utilise the heat produced. However, for hygienisation purposes, the liquid composting system is probably preferable, since the temperature is higher during a longer period of time. Which of these two systems to choose is a matter of the conditions at the relevant site. Factors influencing this are: the need of hygienisation before spreading the residues on farmland, access to renewable energy sources such as wind-power electricity and biofuels, and the potential to utilise the energy produced (heat or gas). In addition, the type of energy sources substituted by utilisation of the energy produced is of vital importance when choosing a system. It is of course best to choose a system that decreases the consumption of the fossil energy sources used today.

The heavy metal contamination of arable land due to recycling of organic residues was mainly an effect of spreading the manure from the 15 dairy cows and their young stock included in the calculations. The contribution from household residues was small, but might still be of importance. Concerning the household residues, the results obtained in this study indicated that, in general, no more heavy metals were recycled when more plant nutrients were recycled since many of the metals came with household organic wastes, which were recycled in all scenarios. The exception was mercury (Hg), which was found in faeces (probably originating from dental amalgam, which should disappear over time as other dental materials are used nowadays). However, the results concerning heavy metals have to be carefully interpreted since the quality of the data can be questioned. In practice, the metal content in different fractions of organic waste may vary a lot, especially in the case of copper. Regions with hard water generally have high levels of copper in sewage sludge. This copper originates from hot-water pipes used in houses. One possible long-term solution discussed previously is blackwater separation. It is an expensive, but in the long run maybe necessary, investment to obtain sufficiently clean organic fertilisers. The greywater would then have to be dealt with separately.

Food production and processing

For the other half of the cyclic system, the food production and processing part, bread production was closely studied and liquid milk production was the subject of a preliminary study. These are only two parts of the complex food-production system but they represent two rather different kinds of staple foods. The study of these two systems gave results that pointed in the same direction, so I think it is possible to draw some general conclusions.

The first is that *small-scale food-production systems <u>can</u> have advantages from an environmental impact and energy consumption point of view*, but that small-scale does not of its own mean "environmentally friendly". The simulations performed showed that an energy-optimised large-scale system was better, in these aspects, than a non-optimised small-scale system. Furthermore, although the energy-optimised small-scale processing plant was assumed to be somewhat more energy consuming compared to energy-optimised large-scale processing plants, the entire small-scale system showed better results concerning most of the impacts categories included. This was especially valid for environmental impacts, while for energy consumption the difference was smaller. The two most essential parts of the systems were the processing and the transport of food from the shop to the consumer's home by private car. Emissions generated in the long-distance transport also contributed to environmental impacts but seemed to be relatively smaller than the private car transport. This is of course very dependent on the choices of transport distances chosen in the study.

In any case, the second conclusion that can be drawn for food-production systems is that: *if small-scale systems are to have smaller environmental impacts, they have to take advantage of the small-scale throughout the entire system*, meaning that not only the processing, but also the distribution system should be small-scale. Thus the local market should be close enough to make it possible for consumers to walk or to use a bicycle etc. when shopping, all in order to minimise private car transport.

Efficient home-distribution systems, similar to the milk-bottle system common in e.g. England, might be another option to lower environmental impacts. In principal, this type of distribution can be used by food-processing companies of all scales. Small-scale processors may manage such a system themselves, while on the large-scale this system would probably be administered by retailers, possibly by Internet shopping-systems. This area, which it was not possible to include in this study, is of utmost importance when discussing food systems and environmental impacts. More research effort needs to be put in here.

Another aspect that may be positive for small-scale processing is the ever decreasing cost of process-regulation technology. As it becomes cheaper, it should be more available for small-scale implementations, thus making them more economically competitive. A shift in taxation from taxes on labour to taxes on natural resources would probably also favour the small-scale, since transport would then become more expensive. On the other hand, energy consumed in processes would also become more expensive, which could be negative for small-scale processing since generally more energy is consumed in this part of the system compared to large-scale facilities. However, the stationary type of energy consumption is easier to control than the spread-out consumption in vehicles, so there might be differences in levels of taxation (in favour of the stationary type).

What might be the largest problem for small-scale processing to deal with would be to keep the energy efficiency high in every part of the system; pre-heating-time and full-temperature-time of heating furnaces, use of warm water, ventilation, etc. This might be hard to combine with production of many types of products, since the internal flow of products gets very complicated and thus hard to manage logistically. A diversified production is probably necessary if one want to have a local market with short transport distances and a good assortment of products.

Cyclic food systems

When studying the examples of the whole food-system presented in this thesis, one can conclude that the small-scale cyclic system needed only a third of the energy compared to the large-scale conventional system. Dominating here was the organic waste management system (including agriculture), while the bread production system stood for a minor part. Aggregating processing for many different foods would of course change this relationship. It should also be remembered that this analysis only included a few parts of the agricultural sector. For example animal farming, which is most probably a major contributor to environmental impacts and resource consumption, was not included here. It was assumed that the parts omitted did not change between the scenarios studied. That seems reasonable, but – it makes it impossible to decide the importance of differences between scenarios concerning environmental impact in the whole food system from results obtained here. For this, studies of the agricultural processes involved have to be performed. A recently published LCA study of primary milk production (Cederberg, 1998) indicates that global warming, acidification and eutrophication impacts could be much larger in the agricultural production of milk than in processing and distribution (Table 16 compared to Table 14), while photo-oxidant formation and energy consumption could be in the same order of magnitude in both systems.

Table 16. Environmental impacts and energy consumption for milk primary production. *(Cederberg, 1998)*

	Impact per kg milk
Global warming, GWPs 100 years *(kg CO_2-equivalents)*	0.94 - 0.98
Acidification, max scenario *(mmol H^+)*	0.50 - 0.56
Photo-oxidants, POCPs *(g ethene-equivalents)*	0.28 - 0.36
Photo-oxidants, NO_x *(g NO_x)*	1.8 - 2.1
Eutrophication, max scenario *(g O_2-equivalents)*	260 - 290
Energy consumption *(MJ)*	2.4 - 2.8

To be able to compare the environmental impacts for the primary production of milk with the impacts for processing and transport of milk studied here, some recalculations had to be performed. The figures in Table 16 were multiplied by 15 000 (150 kg milk per person and year, calculation from Becker, 1992) giving the environmental impacts for primary milk production required for 100 people and year (Table 17). Please note the differences in units. Comparing these figures to the ones presented in Table 15 gave a somewhat different picture. Almost all impact categories were in the same order of magnitude (at least for the Cyclic-Small-Scale-scenario), indicating that both the animal production and the parts of the food system studied in this thesis should be of importance for environmental impacts and energy consumption. Comparing with the Situation-of-Today-scenario, indicates that the processing and distribution are larger contributors to NO_x-emission, eutrophication and energy consumption than the primary production of milk.

Table 17. Environmental impacts and energy consumption for milk primary production.
(calculated from Cederberg, 1998 and Becker, 1992)

	Impact per 100 people and year
Global warming, GWPs *(kg CO_2-equivalents)*	14 100 - 14 700
Acidification, max *(mol H^+)*	7.5 - 8.4
Photo-oxidants, POCPs *(kg ethene-equivalents)*	4.2 - 5.2
Photo-oxidants, NO_x *(kg NO_x)*	27 - 32
Eutrophication *(kg O_2-equivalents)*	3 900 - 4 350
Energy consumption *(MJ)*	36 000 - 42 000

An aspect not included in this study is product quality. It is a vast subject that includes many different issues like taste, smell, look, freshness, price, hygienic properties, storage properties, nutritional value, non-toxicity, trust (in a brand or personal relationship to producer/processor etc.), packaging, knowledge about the producer, knowledge about harvest or production date/time, knowledge about "animal-friendly" production, etc. etc. I believe that many, but not all, are in the favour of local small-scale systems.

Another aspect that may be of importance is the consumption of materials and environmental impacts for the construction of infrastructure, vehicles, technical equipment, buildings etc. Concerning equipment and buildings, it is easy to believe that large-scale plants are better. For infrastructure, the small-scale systems should definitely be more favourable. However, to assume that we would manage with smaller or fewer roads if we chose to have small-scale systems would probably be to go too far, but this discussion and scientific field should be further developed (by others). At least, it can be concluded that transport was a significant contributor to the environmental impacts included in the analysis, although it was not of any large importance concerning energy consumption. This indicates that systems using less transportation are to prefer. Furthermore, transports contribute largely to environmental impact categories not included in this study, like noise, risk for accidents, toxicological impacts on humans, land use, habitat alterations and influence on biological diversity.

Computerised mathematical models as a tool

The models developed in this project should be possible to use as a tool in planning of new settlements, or in assessment and development of existing structures. They are, however, not easy to use by whoever is interested. Firstly, they are not constructed to simply put some figures in and get some results out, though in principal it is possible to develop them to such a level. Secondly, their structure and many underlying assumptions make them too complex to be used without educated "consultants". Of course, these models can be run by many people with some experience in computers, but then they would not produce reliable results.

Conclusions

- No system is "the best" concerning all environmental impact categories
- Different systems may be appropriate at different places or in different situations, as the environmental effects may vary greatly between localities
- Different systems may be appropriate at different places and in different situations, due to the energy sources which are available and the energy sources which are substituted
- Different systems may be appropriate at different places or in different situations, as the purpose the system has to fulfil may vary a lot
- Amounts and environmental impacts of organic wastes from households are generally small compared to internal flows of manure etc. in agriculture
- If organic wastes are to be used as fertilisers it is important to remember and keep track of their properties as fertilisers
- Blackwater sorting systems are a means to increase plant nutrient recycling rate and thus grain yields
- Urine separation increases nitrogen recycling rate and thus grain yields
- Urine separation lowers the energy consumption
- One scenario, with anaerobic digestion treatment (with no hygienisation of incoming material) and a urine-separating toilet system, showed results which indicated that the organic waste management system and agriculture could be energy self-supporting
- Small-scale food-production systems *can* (but does not necessarily) have advantages from an environmental impact and energy point of view
- To really utilise the advantages small-scale systems may have, they should be combined with a nearby local market in order to minimise *all* transport
- An energy-optimised large-scale system is better, concerning environmental impact and energy consumption, than a non-optimised small-scale system
- The two most essential parts in the food-processing systems studied here were the processing itself and the private-car transport of food from shop to consumer’s home
- Electricity produced as Swedish average (mainly hydropower and nuclear power) is clean concerning the environmental impact categories included in this study, but consumes much primary energy resources – thus, it is best used for purposes other than heating (where it can more easily be substituted for resource conserving and less environmentally disturbing alternatives)
- Transport has a large environmental impact but is not that important concerning energy consumption in the systems perspective

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Appendices

Appendix 1. Sludge separator and infiltration model data

% This procedure initiates the MATLAB vectors used by the Sludge separator and Infiltration sub-models.

% The %-symbol indicate that what follows are comments used only for explanation

% References to "Hulta" are unpublished data from measurements of single houses sludge separators in Hulta, a small village outside Linköping in the south of Sweden.

% DM = dry matter

ssOrgRed=0.10;	%A proper designed sludge separation construction reduces BOD/COD with 10-20%. Små avloppsanläggningar. SwEPA 1987. Page 12.
ssDMRed=0.10;	%Calc. from amount DM in sludge per person in Hulta and amount DM in sewage water (Sundberg 1995). (4.8 kg DM sludge/pers, year) / (63,9 kg DM sewagewater/pers, year * 0,75 share "toilet-visits at home")
ssSSRed=0.70;	%A proper designed sludge separation construction can reduce deposite and suspended material with circa 70%. "Små avloppsanläggningar. SwEPA 1987. Page 12.
ssH2ORed=0.01;	%99% of the water passes through. Calculated from Hulta (800 l sludge/p,år)/(200 l water/p,day*365 days)

ssNutrientsInSludge=[; % COLUMNS NUTRIENT CONTENT IN SLUDGE kg/kg DM

50e-3;	%N-tot in sludge, kg per kg DM (Landers 1995; Andersson 1992; Hulta 1997)
9.5e-3;	%N-NH3 in sludge, kg per kg DM (Andersson R, 1992; Hulta 1997).
8e-3;	%P in sludge, kg per kg DM (Andersson 1992; Landers 1995; Hulta 1997)
5e-3;	%K in sludge, kg per kg DM (Andersson 1992).
17e-3;	%Ca in sludge, kg per kg DM (Andersson 1992).

ssMetalsInSludge=[; % COLUMNS HEAVY METAL CONTENT IN SLUDGE kg/kg DM

14e-6;	%Pb	(Andersson 1992; Hulta 1997).
0.8e-6;	%Cd	(Landers 1995; Hulta 1997).
0.67e-6;	%Hg	(Landers 1995; Hulta 1997).
500e-6;	%Cu	(Landers 1995; Hulta 1997).
8.3e-6;	%Cr	(Landers 1995; Hulta 1997).
8.9e-6;	%Ni	(Landers 1995; Hulta 1997).
600e-6;	%Zn	(Andersson 1992; Hulta 1997).

```
ssInfiltration=[ ;      % COLUMNS REDUCTION OF THE SINGLE FRACTIONS IN
                        THE INFILTRATION, PART LEFT IN SOIL
0.75                 ;% C-tot, SwEPA 1987.
0.9                  ;% BOD7, reduction can be as high as 90 to 95 %. SwEPA 1987.
0.2                  ;% N-tot, SwEPA 1987.
0.2                  ;% N-/NH4 (ammonium largely converted to nitrate) SwEPA 1987.
0.2                  ;% N-NO3
0.6                  ;% P, SwEPA 1987.
0.9                  ;% Particles/Susp. mtrl, SwEPA 1987.
0.8                  ;% COD, SwEPA 1987.
```

Appendix 2. Liquid compost and peat filter model data.

%This procedure initiates the vectors and constants used by the liquid compost model.
%The %-symbol indicate that what follows are comments used only for explanation

```
lcElConsPerM3=100          ;%el cons. MJ/m3, (94 MJ/m3, Sæther 1997)
lcHeatProdPerM3=150        ;%heat production MJ/m3, (180-200 MJ/m3 Norin 1996)
                           %(Skjelhaugen & Sæther, 1994 gives ca 70-90 MJ/m3)
lcDegrade=[ ;     % Columns the PART of the input slurry that is left after the
                  degradation.
                  % Data available for VS, DM, N-tot, N-NH3 and COD only
0        ;% C-tot, all C parts are diminished but no data available
0.83     ;% VS, calculated from Skjelhaugen&Donantoni 1997 (0,60 Norin -96)
0.85     ;% DM, calculated from Sæther 1997 (0,65 Norin -96)
1        ;% H2O
0.999    ;% N-tot, 0,13 % of N-tot goes of as ammonia, Sæther 1997.
               %Norin 1996 gives 4%.  The difference reflects differences in air flows.
1.07     ;% N-NH3/NH4, calculated from Sæther 1997
               %(increase due to degradation of org. mtrl)
1        ;% S-tot
1        ;% P      Norin -96
1        ;% Cl
1        ;% K      Norin -96
1        ;% Ca
1        ;% Pb
1        ;% Cd
1        ;% Hg
1        ;% Cu
1        ;% Cr
1        ;% Ni
1        ;% Zn
0.73     ;% COD, calc. from Skjelhaugen&Donantoni 1997 (0,45 Norin -96)

%Biofilter  %%%%%%%%%%%%%%%%%%%%%%%%%%%%
lcPeatDMPerM3=0.03    ;%kg DM spagnum peat per m3 slurry, calculated from Sæther
                      1997. (75,2 kg peat / 638,75 m3 * 0,235 kg dm per kg peat =
                      0,028 kg dm per m3)
lcFilter=[ ;          % Columns the part of gases NOT caught in the biofilter
0                     ;% N-tot, N-tot = NH3
0                     ;% N-NH3/NH4
                         0% of NH3 goes through the filter, Sæther 1997. (2%, Norin 1996)
lcPeat=[ ;            % Columns the peat used in biofilter,  kg / kg DM
                      %All figures from Svensson, 1997
```

```
0.025           ;% VS
11400e-6        ;% N-tot
371e-6          ;% P
200e-6          ;% K
2590e-6         ;% Ca
11.3e-6         ;% Pb
0.2e-6          ;% Cd
0.1e-6          ;% Hg
20.8e-6         ;% Cu
2.1e-6          ;% Cr
1.4e-6          ;% Ni
24.4e-6         ;% Zn

%Residue tank%%%%%%%%%%%%%%%%%%%%%%
NEmissionStorage = 0.01     ;% Part of incoming NH4 emitted as NH3 during 6 months
                            %storage in a covered lagoon, 10 % if not covered.
CH4EmissionStorage = 0.05;% Part of incoming VS emitted as CH4.
lcResidueTank=[ ;           %Columns the part of the residue that is left after storage in
                            45 weeks. All figures from Skjelhaugen & Donantoni 1997.
                            Data available for VS, DM, N-tot, N-NH3 and COD only
                            Losses multiplied with 26/45 to get six months storage.
0.98    ;% VS   S&D give 2.8%losses for 45 weeks
0.99    ;% DM   S&D give 1.6%losses for 45 weeks
1       ;% H2O
0.91    ;% N-tot        S&D give 16%losses for 45 weeks
0.88    ;% N-NH3/NH4    S&D give 20%losses for 45 weeks
1       ;% S-tot
1       ;% P
1       ;% Cl
1       ;% K
1       ;% Ca
1       ;% Pb
1       ;% Cd
1       ;% Hg
1       ;% Cu
1       ;% Cr
1       ;% Ni
1       ;% Zn
0.87    ;% COD S&D give 12.5%losses for 23 weeks
```

Appendix 3. Tractor and spreading model data.

```
%This file initiates the spreading submodel
%The %-symbol indicate that what follows are comments used only for explanation

%-------------Cultivation and Field Data ----------------------------------------
sgDistStore2FieldKm=2          ;%distance store to field in km
sgHoSpreFieldAreaHa=2.0        ;%ha. Field area, ha.
sgHoSpreFieMaxWidm= 100        ;%m. Maximum field width in m.
sgHoSpFieClayPerc=30           ;%Percent clay content. ASSUMED.
sgHoTopSoilH2OCont=3.8         ;%ASSUMED Water content when spreading
sgHoSubSoilH2OCont=4.0         ;%ASSUMED Water content when spreading
sgFieldFormRatio=sgHoSpreFieMaxWidm/(sgHoSpreFieldAreaHa*100);

%-------------Hose Spreader  ----------------------------------------
sgDryHiLim=0.9      ;% Products with water content above this is spread with hose
                    spreader, otherwise with dry spreader
sgHoSpreEmptWeigKg=4000   ;%kg
sgHoSpreTankVLitre=10000   ;%
sgHoSprePumpLpH=210000    ;%Pump capacity influences loading time (=3500 l/min)
sgHoSpreTrVelKmpH=15      ;%Velocity to and from the field with hose spreader
sgHoSpreTrAddTime=0.05    ;%Additional time in hours per load for transport to and
        from the field. 0.033 (2 min/load) for 8 and 10 ton, 0.023 for 5 ton spreaders
        calc. from Elinder&Falk. A hose ramp might take one more minute
sgHoSpreadPTO=0           ;% Power from PTO, for direct use
sgHoSprePumpHgtmFi=5      ;%Total pressure height in m when loading spreader.
sgHoSpreDistrkW=8.5       ;% Effect for distribution of the slurry, (hydraulic effect)
sgHoSprePumpHgtmSp=5      ;%Total pressure height in m when spreading.
sgHoSpreNH3P2SatPre=1     ;%Ratio between actual pressure and saturation pressure
sgHoSpreNH3EmiPerc=8      ;%Ammonia emission after spreading in percent of NH4+.
                    Choose one figure from below:
                    5-7% :Spring operation, worked into soil after 1 h. (1st value urine,
                    2nd liquid manure)
                    10-20% :Spring operation, worked into soil after 12 h
                    3-10% : Autumn operations, worked into soil after 1 h
                    15-25% :Autumn operations, worked into soil after 12 h
sgHoSpreFieVelKmpH=5      ;%  km/h. Work velocity in the field.
                    NOTE, pump capacity limits!! Manually calculated. Tabulated data
                    in Elinder&Falk for liquid manure spreading are  8 and 10 km/h
sgHoSpreWorkWidthm=12         ;%m. Work width of the spreader
sgHoSpreFieAddTPerc=75        ;%In percent of effective work time. Additional time
                    for field spreading. According to Elinder & Falk 60% should be
                    used for working widths below 6m and 75% for working widths
                    above 6m for liquid manure spreaders.
```

sgHoSpreTurnTiHpHa=0.02 ;%hours/ha. Time for turnings
In Elinder & Falk 0.03 hours is used for liquid manure spreaders with work width up to 4m and 0.02 above 6m work widths.

sgHoSprePassesNo=1 ;%Number of operations by the spreader per year

sgHoSpAxTirePkPaVec=[80, 80, 100, 100] %Column vector with the pressure in the different axles

sgHoSpAxWeiKgVec=[2900, 4000, 4000, 4000] %Column vector with the axle loads of the different axles. Valmet 8450 with OMAS MBB100 spreader + 800 kg for the hose-ramp

sgHoSpreAWeiFactVec=[1 1 1 1] ;%How the different axles should be weighed in the subsoil. Axels with linearly changing weight be given as full and empty, each with weigh 0.5

urHoSpreVentAir2Liq=1 ;%Volume of the ventilated air to the liquid

%------------dry spreader---

sgDrySpreEmptWeigKg=4000 ;%kg. Hilt HS11-2000

sgDrySpreLoadKg=8000 ;%kg. Hilt HS11-2000

sgDrySpreLoadTime=0.1 ;%(guess)(h/load) 12 min=0.2 tim, 5 min = 0.083,
%DATABOKEN gives 12 ton/h (40 min) for tractor with front loader. 25 ton/h for wheel-loader

sgDrySpreFieVelKmpH=6 ;% km/h for spreading dry products

sgDrySpreNH3EmiPerc=15 ;%Ammonia emission after spreading in percent of NH4+. Choose one figure from below:
50% :Spring operation, worked into soil after 12 h
20% : Autumn operations, worked into soil after 1 h
50% :Autumn operations, worked into soil after 12 h

sgDrySpreHydrkW=7 ;% Hydraulic power (kW) for the bottom conveyor

sgDrySpreadPTO=10 ;% kW Power needed for the rollers

sgDrySpreTrAddTime=0.033 ;%Additional time in hours per load for transport to and from the field. 0.033 (2 min/load) for 8 and 10 ton, calculated from Elinder&Falk

sgDrySpreTrVelKm=15 ;%Velocity to and from the field with dry spreader

sgDrySpreWorkWidthm=12 ;%m. Work width of the spreader

sgDrySpreFieAddTPerc=60 ;% As above, for dry spreaders above 3 tons load, add time is 60%

sgDrySpreTurnTiHpHa=0.02 ;% Not mentioned in Elinder & Falk, same value as for hose spreaders is used

%---------------Tractor ---

sgDrySpreTracWeigKg=5900 ; %Valmet 8450

sgHoSpreTracWeigKg=5900 ;%Valmet 8450

sgFrontLoaderMJPerKg=0.002 ;%ca 0.05 litre/ton, Calc. from Volvo Wheel Loader AB. 35.8 MJ/l diesel.

%-----------Emissions --------------------------------------

%trEmissionsAB from transport initialisation file (Appendix 4).

Appendix 4. Transport model data; bread production study.

%This file initiates the transports with truck and trailer, ordinary truck, private car, and tractor with wagon

%This version used in large-scale bread production scenarios

%The %-symbol indicate that what follows are comments used only for explanation

%-------------TRUCK AND TRAILER (TT) ---

%Trucks and trailers in Bread Production System scenarios

%TT1=Grain transport to mill (in large scale scenarios)

%--------------Distances --

trkmOneWayTT1=210 % Andersson 1998 give 210 km for the two industrial bakeries.

%--------------Loads --

trMaxLoadOnTT1=30; %Tonnes

trReturnTransTT1=0.5; % If the truck goes empty back insert 0.5, if the truck is full on the return insert 1. I assumed empty return trip, the lorry uses a special body of trailer

%-------------- Fuel consumption ---

trMJPerkmFullTT1=17.8; % 5.0 litre/10 km

trMJPerkmEmptyTT1=10.7; % 3.0 litre/10 km

%--------------------- ORDINARY TRUCK (OT) -----------------------------------

%Trucks in Bread Production System scenarios

%OT1=Plastics and paper to incineration

%OT2=Bread distribution, regional

%OT3=Bread distribution, long-distance

%------------------ Distances --

trkmOneWayOT1=18 %Tillman et al. 1991 give 18

trkmOneWayOT2=350 %Andersson 1998 give 347 km for Industry 1 (calc. from) and max 200 km for Industry 2

%Eskelunds average 202, aver. town only 113

trkmOneWayOT3=260 %Andersson 1998 give 260 km for the Industry 1

%------------------ Loads --

%Tillman 1994 give total truck weights (truck+load): city distrib. 14 tonnes, regional 24 t

trMaxLoadOnOT1=8 ; %Tonnes.

trMaxLoadOnOT2=4 %Tonnes. Eskelunds average 0.58, aver. town only 0.64 (van, lower energy cons.!)

trMaxLoadOnOT3=13.3; %Tonnes. Maxload long-distance trp 19 tonnes, loadfactor 70% (Andersson 1998)

trReturnTransOT1=0.5; %If the truck goes empty back, insert 0.5, if the truck is full on the return, insert 1

trReturnTransOT2=1; %Use 1 for the modified bread distribution model (transport distances are full-trip-figures) trReturnTransOT3=0.7; %Loadfactor 70% (Andersson 1998).

%-------------- Fuel consumption --

%Emissions from this set up gives about half of the CO2 and SO2 emissions Tillman 1996 give and about 1/4 for NOx. One possible/probable explanation is the rate of load (48%) Tillman uses and further, for NOx Tillman gives maximum allowed emissions, which is probably higher than in practise.

%trMJPerkmFullOTx=17.8; % 5.0 litres/10 km ord. lorry, city traffic/short transports. "Back-calc." of Tillmans figures

%trMJPerkmFullOTx=16.0; % 4.5 litres/10 km sludge suction tank lorry

%trMJPerkmFullOTx=14.2; % 4.0 litres/10 km "ordinary" lorry (ORWARE and pers. comm. Girma Gebresenbet)

%trMJPerkmFullOTx=12.5; % 3.5 litres/10 km straw transport (straw has less weight)

%trMJPerkmEmptyOTx=10.7; % 3.0 litres/10 km bread distribution load 4-5 tonnes bread (pers. comm. Girma Gebresenbet)

%trMJPerkmEmptyOTx=7.1; % 2.0 litres/10 km empty lorry

%trMJPerkmFullOTx=5.34; % 1.1 litres/10 km Pick-up or van (pers.comm. Girma Gebresenbet)

trMJPerkmFullOT1=17.8; %Collection with many stops and city traffic

trMJPerkmEmptyOT1=7.1;

trMJPerkmFullOT2=10.7

%trMJPerkmEmptyOT2=7.1; not in use, bread distribution average load data

trMJPerkmFullOT3=14.2;

trMJPerkmEmptyOT3=7.1;

%--------------- TRACTOR AND WAGON TRANSPORTS ---------------------------------

trTractorDistOnRoadKm=30 %km

trWagonEmptyWeightKg=3000; %kg.

trLoadKg=8000; %kg.

trLoadTime=0.25; %(hour/load) Loading and/or unloading time when tractor engine runs on low idle rpm. Zero when tractor turned off. 15 min=0.25 h, 12 min=0.2 h

trTrpAddTime=0.033; % Additional time in hours per load for trp from the field 0.033 (2 min/load) for 8 and 10 ton, calc. from Elinder&Falk (for manure spreading)

trTrpVelocityKmh=20; %Driving velocity. Assume 15 for short trp to/from field, 20 for longer road transports.

trTractorWeightKg=5900 ;%Valmet 8450

%------------- CAR (PETROL ENGINE) --

trKmOneWayCar=5

trReturnTransCar=0.5; %If the trip is made exclusively for the shopping (or whatever purpose is modelled) insert 0.5, if the trip fulfils other purposes as well insert 1. For 25% "shopping-allocation" insert 2.

trKgBoughtEachTime=2 %100 kg bread per 3 persons and year (91.5 g/pers&day ;ages 4-74), calc. from Becker 1992. Choose 1 for shopping twice a week and 2 for once a week, with assumptions above.

```
trMJPerKmCar=4.1;      %MJ/km, incl. precombustion (3.85 excl. precomb.)
                       %(Eriksson et al. 1995; city distribution by car)
trAllocFactorCar=0.06; %Part bread of total bought. Bread is about 6% of total food
                       consumed (Calc. from Becker 1992)

%--------------- EMISSIONS FOR ALL A TO B TRANSPORTS ------------------------------
%---------------------------------------------------------------------------------------------------------
%Hansson et al. 1998 indicate different emissions for tractor transports but due to lack of
        data for many substances the data for lorry trps are used for all diesel fuelled trps.

trEmissionsAB=[;%  kg/MJ used diesel fuel (Egebäck&Hedbom 1991, and Grägg 1992)
             (*, complemented with precombustion emissions from Tillman 1994)
21.3e-3      ;% C-tot (12/44 of CO2)
78e-3        ;% CO2 fossil (*) 74+4
1e-6         ;% CH4
0.074e-3     ;% VOC (*) 0.066+0.008 (Tillman give HC = 0.09+0.008)
2.50e-9      ;% PAH
0.29e-3      ;% CO  Tillman give 0.34e-3
0.537e-3     ;% N-tot
0.534e-3     ;% NOx-N (*) 0.53+0.004 ORWARE ; Tillman 1994 give 0.9 which is
                    max. allowed in Sweden
2.6e-6       ;% N2O-N
0.054e-3     ;% S-tot (*)
0.054e-3     ;% SOx-S (*) ORWARE 0.093e-3 (Tillman 0.094+0.014)
13e-6        ;% Particles/Susp. solids  (Tillman give 1e-4 !)

trEmissionsCar=[;% kg/MJ used petrol fuel, incl. precombustion  (Eriksson et al. 1995)
23.2e-3      ;% C-tot (12/44 of CO2)
85.1e-3      ;% CO2 fossil
24.4e-6      ;% CH4
0.649e-3     ;% VOC
6.39e-3      ;% CO
0.307e-3     ;% N-tot
0.305e-3     ;% NOx-N
1.88e-6      ;% N2O-N
15.1e-6      ;% S-tot
15.1e-6      ;% SOx-S (30.2e-6 kg SOx)
5.56e-6      ;% Particles/Susp. solids
```

Appendix 5. Grain drier and mill model data.

%This file initiates the variables and vectors used in grain drying and mill sub-model
and the air emission vectors for heating (used also in the bakery)

%The %-symbol indicate that what follows are comments used only for explanation

%GRAIN DRIER¤¤

% Amount water evaporised is calculated as follows:

% WC(water content, part of 1) = kgH2O / (kgH2O+kgDM) which give
kgH2O=kgDM*WC/(1-WC)

% kgH2Oevaporised = kgH2Oin - kgH2Oout = kgH2Oin -
kgDM*grMoistContentOut/(1-grMoistContentOut)

grGrainH2OCont=0.20 ;% (kg/kg grain) Water content in newly harvested grain

grMoistContentOut=0.14 ;%Moisture content in dried grain

grFuelConsDrier=5.4 ;%MJ, fuel consumption per kg water vaporised (incl. heat
losses). Theoretical minimum 2.26 (Jacubczyk et al. 1987)
Sigurd Regnér uses 0.125 kg fuel / kg water vaporised, which give
5.4 MJ / kg water (pers. comm.)

%Estimation from references below indicate:
6 for m. c. below 20%, 5 for higher m.c. and possibly 4-4.5 for m.c.s around 30%
6.6 for drying 18% to 15% moisture content (Sonesson 1993, from SMP 3080)
GAS FUEL makes it possible with DIRECT HEATING which lowers fuel
consumption by 15-30 % (Regnér,1987) 20% saving of 5.4 give 4.3 MJ/kg water

grElConsDrier=0.3 ;%MJ electricity per kg water vaporised

0.3 for m.c. around or below 20%, for m.c.s around 30% 0.2 or 0.25
SMP 3099 & 3193 (1987 & 1989) official testing laboratory report
MEPU-drier 5,3 - 6,2 MJ/kg water (33% and 21% incoming water content)
5,1-6,1 MJ fuel/kg water; 0,25-0,32 MJ el/kg water
SMP 3080 (Maskinprovningarna 1987) official testing laboratory report
Akron-drier 3,96 - 6,56 MJ/kg water (32,4 and 16,7 % incoming water content)
4,01-6,68 MJ fuel/kg water; 0,16-0,36 MJ el/kg water (valid for wheat)
Incoming water contents at 16-18% for wheat give ca 6,5 MJfuel/kg and 0,34
MJel/kg

%GRAIN MILL¤¤

grFlourExtrRate=0.80 %Flour extraction rate (utmalningsgrad)

grEnergyConsMill=0.36 %MJ el/kg flour LARGE DIFFERENCE BETWEEN
BREEDS AND DIFFERENT WATER CONTENTS
0.11-0.37 for wheat, different flour extraction rates (95 and 72 %);
0.29 78% extr.rate (Jakubczyk et al. 1987)
0.44 calculated from Kent&Evers
Lewicki 1987 give figures ten times higher (probably wrong cited)
Andersson 1998 give 0.32 Industry1; 0.42 Industry2; and about 0.1
Local bakery; 0.45

```
%EMISSIONS FOR HEATING¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤
grEmissOilBurning=[ ;% COLUMNS AIR EMISSIONS DUE TO BURNING OF
                    FOSSIL OIL FUEL (kg emission per MJ fuel)
                    % Habersatter et al. 1998 Table 16.9 page 375 (Heating oil EL)
                    % ALSO USED FOR OIL-HEATED BAKERY OVEN
0.023       ;% C-tot (12/44 * CO2-f)
0.0829      ;% CO2-f
0.98e-4     ;% CH4
3.1e-4      ;% VOC
6.0e-9      ;% CHx
1.6e-6      ;% PAH
2.7e-5      ;% CO
3.65e-5     ;% N-tot
2.0e-9      ;% N-NH3/NH4 (2.4e-9 kg NH3/kg)
3.6e-5      ;% N-NOx (0.118e-3 kg NO2/kg)
5.4e-7      ;% N-N2O (8.4e-7 kg N2O/kg)
0.06e-3     ;% S-tot
0.06e-3     ;% S-SOx (0.12e-3 kg SO2/kg)
0.24e-6     ;% Cl (0.25e-6 kg HCl)
6e-3        ;% Particles/Susp. mtrl

grEmissGasBurning=[ ;% COLUMNS AIR EMISSIONS DUE TO BURNING OF GAS
                    FUEL(kg/MJ used fuel)
                    % Habersatter et al. 1998 Table 16.9 page 375 (Natural gas)
0.017       ;% C-tot
0.063       ;% CO2-f
0.18e-3     ;% CH4
0.19e-3     ;% VOC (Tillman give 1.5e-8 for HC)
4.2e-11     ;% CHx
55e-6       ;% PAH
26e-6       ;% CO
19.4e-6     ;% N-tot
5.1e-9      ;% N-NH3/NH4 (6.2e-9 kg NH3/kg)
19e-6       ;% N-NOx (64e-6 kg NO2/kg)
0.43e-6     ;% N-N2O (0.67e-6 kg N2O/kg)
17e-6       ;% S-SOx (35e-6 kg SO2/kg)
0.38e-6     ;% Cl (0.39e-6 kg HCl)
3.4e-6      ;% Particles/Susp. mtrl
```

```
grEmissBioFuelBurning=[ ; % COLUMNS AIR EMISSIONS DUE TO BURNING OF
                        BIO FUEL (kg/MJ used fuel, incl. precombustion)
                        % Habersatter et al. 1998 Table 16.9 page 375 (Wood)
0.38e-3                 ;% C-tot (12/44 of CO2-f plus 12/28 of CO)
0.20e-3                 ;% CO2-f
0                       ;% CO2-b (should be a value here but it's omitted since not
                        accounted for later on in the calculations)
4.8e-6                  ;% CH4
1.1e-5                  ;% VOC (Tillman give 1e-4 for HC)
1.5e-11                 ;% CHx
1.9e-6                  ;% PAH
0.75e-3                 ;% CO
48e-6                   ;% N-tot
9e-6                    ;% N-NH3/NH4 (11e-6 kg NH3/kg)
39e-6                   ;% N-NOx (1.27e-4 kg NO2/kg)
0.5e-6                  ;% N-N2O (0.74e-6 kg N2O/kg)
13e-6                   ;% S-SOx (26e-6 kg SO2/kg)
1.3e-6                  ;% Cl (1.33e-6 kg HCl)
1.5e-4                  ;% Particles/Susp. mtrl
```

Appendix 6. Bakery and packaging model data.

%This file initiates the variables and vectors used in bakery and packing sub-models.

%The %-symbol indicate that what follows are comments used only for explanation

%References: Andersson (1998) refer to Andersson Karin, 1998a
Eskelunds (1996) refer to Nilsson Jonas, 1996

%Scenario specific choices

%primary package

baPlasticBagChoice=1 % 1 if chosen, 0 if paper bags are chosen
(also possible to choose parts, e.g. 0.5)

baPaperBagChoice=1-baPlasticBagChoice;

%secondary package

baPlasticBoxChoice=0 % 1 if chosen, 0 if cardboard boxes are chosen

baCardboardBoxChoice=1-baPlasticBoxChoice;

%water consumption

baWaterConsPerKgBread=1.4 %liter per kg bread, water for cleaning and ingredient
Andersson (1998): 0.8 Industry2, 1.4 Industry1, 3.0
Local bakery (probably 1.5 in bakery). Eskelunds (1996)
1.45

%fuels

baEmissHeatingOven=grEmissOilBurning; %Air emissions due to heating of bakery
oven %Choose one of grEmissOilBurning, grEmissGasBurning or
grEmissBioFuelBurning

baEmissFuelSteamCons=grEmissOilBurning ;%Air emissions due to heating of
steam %Choose one of grEmissOilBurning, grEmissGasBurning or
grEmissBioFuelBurning

%---

% BAKERY

baAdditives=1.4 %Value of "fluffyness", higher value fluffier bread
(have assumed all additive is water in the model)

100 kg of flour makes 145 kg of bread (Kent&Evers); 4 kg additives, the rest is water
Knut Maroni (enøk 1995) give 1.35
Andersson 1998 give Industry1 1.32, Industry2 1.45, Local bakery 1.62
Eskelunds 1.49 kg bread / kg flour (all different sorts of bread); 13.5% additives

%OVEN, choose ONE of the energy sources and its appropriate energy consumption, set
the other to zero.

baOvenElConsPerKg=0 %MJ electricity per kg bread (incl. preheating and possible
steaming in oven)

1.4 electrical heated tunnel oven, large bakery; 2.0 electrical heated
tunnel oven, small bakery "ineffective prod." (Trägårdh et al. 1979)
3.11 average for small bakery as above (Trägårdh et al. 1979)
1.0 Industry 2 (Andersson 1998)
1.19 Eskelunds Hembageri AB, Visby (1996)

baOvenFuelConsPerKg=1.7 %MJ fuel per kg bread, incl. pre heating of oven

0.8-0.9 gas heated, probably not incl. preheating (Lucas et al.)
1.25 average, incl. steaming(?) and pan, no energy source type given (Kent&Evers)
1.7 Industry1 (gas), 2.3 Local bakery (oil, probably not only oven) (Andersson 1998)
2.73 oil small bakery "ineffective prod." (Trägårdh et al. 1979)

%STEAMING and FERMENTATION

baFermElConsPerKg=0 %MJ electricity per kg bread (heating of fermentation room)
0.36 (lokal bakery, Andersson pers. comm. 1998)
1.09 chill-freeze-ferment-chamber (Eskelunds)
1.94 small bakery (Trägårdh et al. 1979)

baSteamFuelConsPerKg=1 %MJ fuel per kg bread (steam for oven and/or fermentation)
0.80 (might be too high) Industry1, 0.13 Industry2 (Andersson 1998)
1.54 (0.41 fermentation + 1.13 steaming) large bakery big losses in steaming (Trägårdh et al. 1979)

%REMAINING ENERGY CONSUMPTION ("THE REST")

baRestElConsPerKg=0.9 %MJ electricity per kg bread (total remaining el. cons. incl. water heating), water heating energy cons. is subtracted in the model
Industry1 0.27, Industry2 0.4, Local Bakery 0.21 (incl. mill!) (Andersson 1998); 0.39 (Eskelunds 1996); 0.85-0.91 (Trägårdh et al. 1979)

baDivFuelConsPerKg=0; %MJ fuel per kg bread (not used so far)

%WATER HEATING

baWaterTempIn=15 ;% ^{o} Celcius, Temperature of incoming water

baWaterTempHotWater=60;% ^{o} Celcius, Temperature of hot water used for cleaning

baWaterHeatElCons=4.18e-3*1*(baWaterTempHotWater-baWaterTempIn);
%MJ electricity per litre WATER used FOR CLEANING
Q=c*m*(t2-t1) WHERE Q=energy in kJ; c=heatcapacitivity in kJ/kg,K; m=mass in kg; t=temperature in degree Celcius
Heating of 1 kg water from 15 to 60 degrees is: Q=4.18*1*(60-15)=188.1 kJ i.e. 0.1881 MJ/kg H2O
0.031 heating of water in recipe, local bakery (Andersson 1998)

baWaterHeatFuelCons=0; %MJ fuel per kg water (not used in this study)

%¤¤

%PACKAGING

%##########

%PLASTIC BAGS made of low density polyethylene (LDPE)

baPlasticBagKgPerKgBread=0.015;%kg per kg bread;
Arnkvist 1997 (11.5 g industry 2 and 17 g industry 1)

baPlasticBagElCons=9.6 %MJ el. energy per kg PE; Boustead 1993

baPlasticBagThermalCons=79.0 ;%MJ thermal energy per kg PE; Boustead 1993

baHeatValuePlasticBag=43 ;%Heat value, MJ per kg PE;

```
baAirEmPlasticBag=[ ;  % COLUMNS AIR EMISSIONS FROM MANUFACTURING
                      OF  PLASTIC BAG PRIMARY PACKAGES
                      kg emission per kg plastic bag (Boustead 1993)
0.34                  ;% C-tot (12/44*(CO2-f))
1.25                  ;% CO2-f
0.021                 ;% VOC
0.9e-3                ;% CO
3.6e-3                ;% N-tot
3.6e-3                ;% N-NOx (12e-3 kg NOx)
4.5e-3                ;% S-tot
4.5e-3                ;% S-SOx (9e-3 kg SO2)
68e-6                 ;% Cl (70e-6 kg HCl)
0.003                 ;% Particles/Susp. mtrl

baWaterEmPlasticBag=[ ; % COLUMNS WATER EMISSIONS FROM
                      MANUFACTURING OF PLASTIC BAG PRIMARY PACKAGES
                      kg emission per kg plastic bag (Boustead 1993)
3.9e-6                ;% N-NH3/NH4 (5e-6 kg NH4)
1.1e-6                ;% N-NO3 (5e-6 kg NO3)
1.6e-6                ;% P (5e-6 kg PO4)
1.5e-3                ;% COD

%###############
%PAPER BAGS made of craft paper
baPaperBagKgPerKgBread=0.021; %kg per kg bread; Arnkvist 1997
baPaperBagElCons=15             ;%MJ per kg paper; Habersatter et al. 1998
baPaperBagThermalCons=55        ;%MJ per kg paper; Habersatter et al. 1998
baHeatValuePaperBag=17          ;%Heat value, MJ per kg paper;
baAirEmPaperBag=[ ;%   COLUMNS AIR EMISSIONS FROM MANUFACTURING
                      OF  PAPER BAG PRIMARY PACKAGES
                      kg emission per kg paper bag  (Habersatter 1998; Table 12.32)
0.29                  ;% C-tot (12/44 of CO2-f plus 12/28 of CO)
1.08                  ;% CO2-f
2.43e-3               ;% CH4
3.65e-3               ;% VOC
22.7e-9               ;% CHX
8.95e-6               ;% PAH
0.98e-3               ;% CO
5.13e-3               ;% N-tot
24.3e-6               ;% N-NH3/NH4
5.13e-3               ;% N-NOx (5.13e-3 kg NO2 per kg paper)
16.6e-6               ;% N-N2O
1.31e-3               ;% S-tot
```

```
1.31e-3                ;% S-SOx (2.62e-3 kg SO2 per kg paper)
23e-6                  ;% Cl (23.6e-6 kg HCl)
1.79e-3                ;% Particles/Susp. mtrl

baWaterEmPaperBag=[ ;  % COLUMNS WATER EMISSIONS FROM
                       MANUFACTURING OF PAPER BAG PRIMARY PACKAGES
                       kg emission per kg paper bag  (Habersatter 1998; Table 12.32)
7.7e-6                 ;% N-NH3/NH4 (9.95e-6 kg NH4)
55.1e-6                ;% N-NO3 (244e-6 kg NO3)
1.2e-6                 ;% P (3.65e-6 kg PO4)
11.1e-3                ;% COD

%##############
%PLASTIC CLIPS made of polystyrene (HIPS)
baPlasticClipsKgPerKgBread=0.001          ;%kg per kg bread;
                              Arnkvist 1997 (0.0005 Industry1 and 0.00143 Industry2)
baPlasticClipsElCons=4.8            ;%MJ per kg PS; (Boustead 1997)
baPlasticClipsThermalCons=85.9   ;%MJ per kg PS; (Boustead 1997)
baHeatValuePlasticClips=40                ;%Heat value, MJ per kg PS;
baAirEmPlasticClips=[ ;%   COLUMNS AIR EMISSIONS FROM MANUFACTURING
                       OF  PLASTIC CLIPS PRIMARY PACKAGES
                       kg emission per kg plastic clips (Boustead 1997)
0.8                    ;% C-tot (12/44 of CO2-f)
2.8                    ;% CO2-f
0.011                  ;% CH4
0.0038                 ;% VOC
0.0002                 ;% PAH
0.0012                 ;% CO
10e-6                  ;% H
0.0036                 ;% N-tot
0.0036                 ;% N-NOx, 12e-3 kg NOx (antar NOx = NO2)
0.006                  ;% S-tot
0.006                  ;% S-SOx, 12e-3 kg SO2
34e-6                  ;% Cl (35e-6 kg HCl)
0.002                  ;% Particles/Susp. mtrl

baWaterEmPlasticClips=[ ;           % COLUMNS WATER EMISSIONS FROM
                       MANUFACTURING OF PLASTIC CLIPS PRIMARY
                       PACKAGES  kg emission per kg plastic clips (Boustead 1997)
6.2e-6                 ;% N-NH3/NH4 (8e-6 kg NH4)
0.5e-6                 ;% N-NO3 (2e-6 kg NO3)
0.36e-3                ;% COD
```

```
%############
%PLASTIC BOXES for distribution of bread made of high-density polyethylene (HDPE)
baPlasticBoxKgPerKgBread=0.414; %kg per kg bread;
                    Arnkvist 1997 (when used one time)
baPlasticBoxElCons=5.79         ;%MJ per kg plastic; Boustead 1993
baPlasticBoxThermalCons=75.2    ;%MJ per kg plastic; Boustead 1993
baHeatValuePlasticBox=43        ;%Heat value, MJ per kg plastic;
baPlasticBoxTimesUsed=500       ;%How many times the box is used in its lifetime

baAirEmPlasticBox=[ ;  % COLUMNS AIR EMISSIONS FROM MANUFACTURING
                    OF PLASTIC BOX SECONDARY PACKAGES (Boustead 1993,
                    Table 20)  kg emission per kg plastic box
0.26                ;% C-tot (12/44 of CO2-f)
0.94                ;% CO2-f
0.021               ;% VOC
0.6e-3              ;% CO
1e-6                ;% H
0.003               ;% N-tot
0.003               ;% N-NOx (0.010 kg NOx)
0.003               ;% S-tot
0.003               ;% S-SOx (0.006 kg SO2)
49e-6               ;% Cl (50e-6 kg HCl)
0.002               ;% Particles/Susp. mtrl

baWaterEmPlasticBox=[ ; % COLUMNS WATER EMISSIONS FROM
                    MANUFACTURING OF  PLASTIC BOX SECONDARY
                    PACKAGES (Boustead 1993, Table 19)  kg emission per kg plastic
                    box
7.8e-6              ;% N-NH3/NH4 (10e-6 kg NH4)
2.3e-6              ;% N-NO3 (10e-6 kg NO3)
0.3e-6              ;% P (1e-6 kg PO4)
0.2e-3              ;% COD
```

%CARDBOARD BOXES for distribution of bread (corrugated cardboard)

baCardboardBoxKgPerKgBread=0.117 ;%kg per kg bread; Arnkvist 1997

baCardboardBoxElCons=5.47 ;%MJ per kg cardboard;
Habersatter et al. 1998 (table 12.109)

baCardboardBoxThermalCons=36.08 ;%MJ per kg cardboard;
Habersatter et al. 1998 (table 12.109)

baHeatValueCardboardBox=17 ;%Heat value, MJ per kg cardboard;

baAirEmCardboardBox=[; % COLUMNS AIR EMISSIONS FROM MANUFACTURING OF CORRUGATED CARDBOARD BOX SECONDARY PACKAGES; kg emission per kg cardboard box Habersatter et al. 1998 (table 12.93)

0.18	;% C-tot (12/44 of CO2-f)
0.644	;% CO2-f
1.09e-3	;% CH4
2.21e-3	;% VOC
26e-9	;% CHX
5.75e-6	;% PAH
0.755e-3	;% CO
6e-9	;% Phenols
1.065e-3	;% N-tot
128e-6	;% N-NH3/NH4 (156e-6 kg NH3 per kg cardboard)
920e-6	;% N-NOx (3.01e-3 kg NOx cardboard)
17e-6	;% N-N2O (0.0265e-3 kg N2O per kg cardboard)
1.4e-3	;% S-tot
1.4e-3	;% S-SOx (2.81e-3 kg SO2 cardboard) (Tillman 0.29)
23e-6	;% Cl (23.3e-6 kg HCl)
0.64e-3	;% Particles/Susp. mtrl

baWaterEmCardboardBox=[; % COLUMNS WATER EMISSIONS FROM MANUFACTURING OF CORRUGATED CARDBOARD BOX SECONDARY PACKAGES; kg emission per kg cardboard box Habersatter et al. 1998 (table 12.93)

8.1e-6	;% N-NH3/NH4 (10,4e-6 kg NH4)
0.3e-3	;% N-NO3 (1.52e-3 kg NO3)
0.9e-6	;% P (2.8e-6 kg PO4)
11.1e-3	;% COD

Appendix 7. Packaging waste handling model data

```
%This file initiates the vectors and variables used in the packaging waste management
   sub-system (incineration and recycling)
%The %-symbol indicate that what follows are comments used only for explanation

waIncinerationChoice =0        ;% 1 if incineration is chosen, 0 if recycling is chosen
                               (also possible to choose parts)
waRecyclingChoice=1-waIncinerationChoice
waRecPlastAllocFactor=0.5      ;%Part of emissions and energy prod./cons. allocated to
                               the recycled plastics. When mtrl recycled once 0.5, when
                               recycled 4 times 0.2, etc. (1/(x+1))
waRecPaperAllocFactor=0.2      ;%Part of emissions and energy prod./cons. allocated to
                               the recycled paper
hhDryPaperTsPart=0.88          ;% DM content in paper
hhCardboardTsPart=0.79         ;% DM content in cardboard
hhMixPlasticTsPart=0.95        ;% DM content in plastics

%¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤¤
%RECYCLING
%NOTE! Internal energy value of the material included in thermal energy consumption!

%PLASTICS (assumed that LDPE, HDPE and PS are equivalent; figures for LDPE)
%Figures are calculated as the difference between recycling and landfilling scenarios in
                               Tillman et al 1991. I.e. transports are included!
%Negative values imply that recycling lowers the energy consumption or the emission
                               compared to the use of virgin material
waPlasticsRecycleElCons=-2.9             ;% MJ el. per kg recycled material
waPlasticsRecycleThermalCons=-38.4     ;% MJ thermal energy per kg recycled
                               material
waPlasticsRecycleEmissions=[     ;% kg emission per kg recycled material
-0.17          ;% C-tot (12/44 * CO2-f)
-0.61          ;% CO2-f
-7.1e-3        ;% VOC
-0.24e-6       ;% Phenols
-0.21e-3       ;% N-tot
-0.21e-3       ;% N-NOx
-0.27e-3       ;% S-tot
-0.27e-3       ;% S-SOx (-0.5439e-3 kg SO2/kg)
-0.05e-3       ;% Particles/Susp. mtrl
```

```
%PAPER PRODUCTS (assumed that recycling of all paper (craft paper and cardboard)
                        is equivalent to corrugated board
%Figures are calculated as the difference between recycling and landfilling scenarios in
                        Tillman et al 1991. I.e. transports are included!
%Negative values imply that recycling lowers the energy consumption or the emission
                        compared to the use of virgin material

waPaperRecycleElCons=-0.5            ;% MJ per kg recycled material
waPaperRecycleThermalCons=-22.2      ;% MJ per kg recycled material
waPaperRecycleEmissions=[            ;% kg emission per kg recycled material
-0.025          ;% C-tot (12/44 * CO2-f)
-0.093          ;% CO2-f
-0.089e-3       ;% VOC
-0.12e-3        ;% CO
-3e-9           ;% Phenols
-0.29e-3        ;% N-tot
-0.29e-3        ;% N-NOx (-0.94e-3 kg NOx/kg paper)
-0.08e-3        ;% S-tot
-0.08e-3        ;% S-SOx (-0.17e-3 kg SO2/kg paper)
-0.49e-3        ;% Particles/Susp. mtrl
```

Appendix 8. Dairy model data.

%This file initiates the variables and vectors used in dairy and packing sub-models.

%The %-symbol indicate that what follows are comments used only for explanation.

%References to dairies used here refer to following personal references given in the references list.

%Arla, Kallhäll (very large dairy in Stockholm)
= Larsson Inger, 1997 (environmental report)

%Kågeröd (small dairy in Skåne, Sweden)
= Hedegård Poul, 1999 (KM-manual ISO 9002 and ISO 14001, and pers. comm.)

%Torsta/Rösta (mini-dairy in Jämtland, Sweden producing mainly cheese)
= Jürss Kerstin, 1999 (pers. comm.)

%Wapnö (on-farm dairy in Halland, Sweden)
= Bengtsson Lennart, 1999 (pers. comm.)

%Scenario specific choices

daMilk=15000; %kg milk per year
(15000 is approx. cons. per 100 persons, calc. from Becker 1992)

%package type

daCardboardChoice=1; % 1 if chosen, 0 if cardboard packages are chosen (also possible to choose parts, e.g. 0.5)

daGlassChoice=1-daCardboardChoice;

%water consumption

daWaterConsPerKgMilk=3 %liter water for cleaning, per kg milk

Arla, Kallhäll (1997) 0.95
Kågeröd (1997) 3.5
Høgaas Eide (1998) about 1 for all three dairies
SwEPA (1991) 1-2
Elsy (1980) small 3.65, middle 2.22, large 2.93, AVERAGE 2.52
Torsta/Rösta (allocated) 4.2
1 litre give water emissions in level with Eide's presented results for large-scale dairy.
2 litre for middle-size and 3 litres for small-size.

%SCENARIOS: LargeBad 2; LargeGood 1; SmallBad 4; SmallGood 3;

%fuels

daEmissFuelCons=fuEmissOilBurning ;%Air emissions due to heating in dairy
%Choose one of fuEmissOilBurning, fuEmissGasBurning or fuEmissBioFuelBurning

%SCENARIOS: LargeBad oil; LargeGood not used; SmallBad oil; SmallGood not used

%DAIRY

daElConsPerKg=0.7 %MJ electricity per kg milk
%DAIRIES USING FUELS FOR HEATING
Kågeröd (1997) 0.33 (+ 0.52 in fuels)
Arla,Kallhäll (1997) 0.196 (0.396 incl. heat from electricity)
Cox & Miller (1985) newer dairies – average
el Australia 0.14 - 0,20 MJ/kg (cartons)
el NZ 0.13 - 0,20 MJ/kg (glass bottles)
Elsy (1980) small 0.13, middle 0.08, large 0.10, AVERAGE 0.09

%DAIRIES USING ONLY ELECTRICITY
Wapnö (1999) 0.79
Torsta/Rösta (1997) 2.6 (total, allocated, hard to assess actually)
Høgaas Eide (1998) small 3.2, middle 1.3, large 0.55 (total, allocated)
Cajander (1996) 0.4 (total, calculated from prognosis)
Verma (1988) 0.92 modern plant glass bottles, 0.50 modern plant one-way containers
Lucas (1985) 0.795 (total), target 0.48 (total)

%SCENARIOS: LargeBad 0.2; LargeGood 0.4; SmallBad 0.4; SmallGood 0.7

daFuelConsPerKg=0 %MJ fuel per kg milk
Arla,Kallhäll 0.199 (produced from el)
Kågeröd (1997) 0.52 (0.50 gas + 0.02 oil)
Cox & Miller (1985) newer dairies – average
fuel Australia 0.22 - 0.46 MJ/kg (cartons)
fuel NZ 0.41 - 0.88 MJ/kg (glass bottles)
Elsy (1980) small 1.24, middle 0.65, large 0.78, AVERAGE 0.85

%SCENARIOS: LargeBad 0.5; LargeGood 0; SmallBad 0.8; SmallGood 0;

%WASTE WATER

%Høgaaas Eide (1998) present results indicating that content of pollutants in wastewater differ between scales. In that study, water consumption did not vary much. However, in order to make the water emissions "proper" it is convenient to adjust the water consumption at top of this file.

%1 litre give water emissions in level with presented results for large-scale. 2 litres for middle-size. 3 for small.

%Data given for Høgaaas Eide (1998) are the once allocated from specific water consumption.

daWaterEmDairy=[; %COLUMNS WATER EMISSIONS FROM DAIRY; kg per kg water

0.8e-3 ;% BOD7
Kågeröd (1997) 1.282e-3
Høgaas Eide (1998) 0.9e-3 large-size dairy, 1.5e-3 middle-size, 2.4e-3 small

50e-6 ;% N-tot ;
Høgaas Eide (1998) 50e-6 large-size dairy, 50e-6 middle-size, 140e-6 small

10e-6 ;% P;
Arla Kallhäll (1997) 12.1e-6; Kågeröd (1997) 66.29e-6;
Høgaas Eide (1998) 7e-6 large-size dairy, 15e-6 middle-size, 27e-6 small

1.5e-3 ;% COD
Arla 2.1e-3;
Høgaas Eide (1998) 1.5e-3 large-size dairy, 2.5e-3 middle-size, 4.0e-3 small

```
%PACKAGING
%##########
%CARDBOARD PACKAGES, "tetras" for milk (liquid packaging board, LPB)
daCardboardKgPerKgMilk=0.028  ;%kg per kg milk
                        (Høgaas-Eide 1998 gives 36 (1-litre?) packages per 1 kg cardboard)
daCardboardElCons=10.97;   %MJ per kg cardboard; Habersatter et al. 1998 (table
                        12.76)
daCardboardThermalCons=42.21;   %MJ per kg cardboard; Habersatter et al. 1998
daHeatValueCardboard=16.81       ;%Heat value, MJ per kg cardboard;
daAirEmCardboard=[ ;     % COLUMNS AIR EMISSIONS FROM
                        MANUFACTURING OF CARDBOARD LIQUID PACKAGES;
                        kg emission per kg cardboard, Habersatter et al. 1998 (table 12.70)
0.09                    ;% C-tot (12/44 of CO2-f)
0.331                   ;% CO2-f
0.444e-3                ;% CH4
1.432e-3                ;% VOC
19e-9                   ;% CHX
4.6e-9                  ;% PAH
0.380e-3                ;% CO
915.028e-6              ;% N-tot
0.228e-6                ;% N-NH3/NH4 (0.277e-6 kg NH3)
910e-6                  ;% N-NOx (2.99e-3 kg NOx)
4.80e-6                 ;% N-N2O (7.54e-6 kg N2O)
0.88e-3                 ;% S-tot
0.88e-3                 ;% S-SOx (1.76e-3 kg SO2) (Tillman gives 0.29)
9.60e-6                 ;% Cl (9.87e-6 kg HCl)
0.0504e-6               ;% Pb
0.0075e-6               ;% Cd
0.00184e-6              ;% Hg
0.461e-6                ;% Ni
0.0965e-6               ;% Zn
0.89e-3                 ;% Particles/Susp. mtrl

daWaterEmCardboard=[ ; % COLUMNS WATER EMISSIONS FROM
                        MANUFACTURING OF CARDBOARD LIQUID PACKAGES;
                        kg emission per kg cardboard, Habersatter et al. 1998 (table 12.70)
6.25e-6                 ;% N-NH3/NH4 (8.03e-6 kg NH4)
0.93e-6                 ;% N-NO3 (4.12e-6 kg NO3)
0.52e-6                 ;% P (1.59e-6 kg PO4)
23.1e-3                 ;% COD

%##########
%GLASS BOTTLES, not in use yet
```

Appendix 9. Soil heavy metal balance – an example.

This calculation shows the amounts of heavy metals both recycled to arable land and the possible export with the grain produced on that land. The grain included was the potential additional grain-production from the organic waste recycling and the base production produced also with no fertilising. The difference between import and export was reported as net soil contamination. However, it should be remembered that the contribution from precipitation and possible other sources was not taken into account. In for example. cadmium contamination of soils in Sweden, precipitation is a major contributor.

The balance calculation in Table 18 shows that Pb, Cu, Cr and Ni were recycled to soil to a higher extent than they were exported from soil. Cd and Hg were about break-even and Zn was exported to a higher extent than it was recycled. However, these results have to be very carefully interpreted since data and model qualities were questionable. The heavy metal content in grain in particular can vary a lot depending on species and cultivar of grain, and the soil where it is grown. Data used here were somewhat old and originated from studies in Finland. Concerning the data for Hg, Cd, and Pb in manure, they were given as "lower than detection limit", which implies that the recycled amounts of these metals were over-estimated. Moreover, it is doubtful if the heavy metal content in grain would be constant when produced on different soils and in different fertilising regimes. I chose to present the results, remembering their weaknesses, as an example of a possible environmental impact. Another relevant comparison that could be made is that against the heavy metal contamination when mineral fertilisers are used. Concerning heavy metals and nitrogen, organic fertilisers will inevitably lose the competition since mineral nitrogen fertilisers contain very close to zero heavy metals. For phosphorus, the comparison is of more interest, but this is not included in the discussion here. Furthermore, of more interest than comparing chemical and organic fertilisers is evaluating whether the organic fertilisers will give net contamination of any importance. This is definitely an area where more research is needed.

Table 18. Heavy metals recycled to and removed from soil due to recycling and grain production emanating from household wastes and wastewater, and manure from 15 dairy cows and their young stock. *(kg per year)*[1]

	Sludge to sewage plant	Sludge to sew. plant Urine sep.	Sludge to agri-culture	Sludge to agri-culture Urine sep.	Comp-ost toilet	Comp-ost reactor Urine sep.	Liquid comp-ost	Liquid comp-ost Urine sep.	Anae-robic diges-tion	Anae-robic digest. Urine sep.
recycled with residues (numbers in brackets are amount in manure, same for all scenarios)										
Pb	0.22 (0.19)[2]	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22
Cd	0.020 (0.019)[2]	0.020	0.020	0.020	0.020	0.020	0.020	0.020	0.020	0.020
Hg	0.003 (0.002)[2]	0.003	0.003	0.003	0.004	0.004	0.004	0.004	0.004	0.004
Cu	5.1 (5.0)	5.1	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2
Cr	0.14 (0.08)	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14
Ni	4.59 (4.56)	4.59	4.59	4.59	4.59	4.59	4.59	4.59	4.59	4.59
Zn	11.1 (10.6)	11.1	11.2	11.2	11.4	11.3	11.4	11.4	11.4	11.4
removed with grain (Koivostinen, 1980)										
Pb	0.03	0.04	0.03	0.04	0.03	0.04	0.03	0.03	0.03	0.04
Cd	0.020	0.021	0.020	0.021	0.020	0.021	0.020	0.021	0.021	0.022
Hg	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.003
Cu	4.1	4.4	4.1	4.4	4.1	4.4	4.1	4.3	4.3	4.6
Cr	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
Ni	1.3	1.4	1.3	1.4	1.3	1.4	1.3	1.4	1.4	1.4
Zn	15.8	17.0	15.8	17.0	16.1	17.1	16.1	16.9	16.9	17.8
net soil contamination										
Pb	0.19	0.18	0.19	0.19	0.19	0.18	0.19	0.18	0.18	0.18
Cd	0.000	-0.001	0.000	-0.001	0.000	-0.001	0.000	-0.001	-0.001	-0.002
Hg	0.000	-0.001	0.000	0.000	0.001	0.001	0.001	0.001	0.001	0.001
Cu	1.1	0.8	1.2	0.9	1.0	0.8	1.1	0.8	0.8	0.6
Cr	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.11
Ni	3.3	3.2	3.3	3.2	3.3	3.2	3.3	3.2	3.2	3.1
Zn	-4.7	-6.0	-4.6	-5.9	-4.8	-5.8	-4.7	-5.5	-5.6	-6.5

[1] Net soil contamination sometimes not the exact difference due to rounding-off errors

[2] Analysis results given as "lower than detection limit"

As the manure stands for the major part of the heavy metals recycled it can be of interest to show the corresponding figures for the household residues part only (Table 19). For Cd, Cu, Ni and Zn many scenarios actually showed negative figures, i.e. more was removed with the grain than imported with the residues. However, the question of data and model reliability stands.

Table 19. Heavy metals recycled to and removed from soil due to recycling and grain production emanating from household wastes and wastewater. *(kg per year)*[1]

	Sludge to sewage plant	Sludge to sew. plant Urine sep.	Sludge to agri-culture	Sludge to agri-culture Urine sep.	Comp-ost toilet	Comp-ost Urine sep.	Liquid comp-ost	Liquid comp-ost Urine sep.	Anae-robic diges-tion	Anae-robic digest. Urine sep.
recycled with residues										
Pb	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
Cd	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001
Hg	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cu	0.16	0.16	0.26	0.25	0.21	0.20	0.21	0.21	0.21	0.21
Cr	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06
Ni	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Zn	0.51	0.48	0.66	0.59	0.77	0.74	0.77	0.77	0.77	0.77
removed with grain (Koivostinen, 1980)										
Pb	0.0003	0.0028	0.0003	0.0028	0.0009	0.0029	0.0030	0.0035	0.0038	0.0040
Cd	0.0002	0.0017	0.0002	0.0017	0.0006	0.0018	0.0018	0.0021	0.0023	0.0024
Hg	0.0000	0.0003	0.0000	0.0003	0.0001	0.0003	0.0003	0.0003	0.0003	0.0004
Cu	0.04	0.34	0.04	0.34	0.12	0.36	0.37	0.44	0.47	0.49
Cr	0.0002	0.0023	0.0003	0.0023	0.0008	0.0024	0.0024	0.0029	0.0031	0.0033
Ni	0.011	0.11	0.01	0.11	0.04	0.12	0.12	0.14	0.15	0.16
Zn	0.14	1.34	0.16	1.34	0.46	1.42	1.44	1.71	1.85	1.93
net soil contamination										
Pb	0.026	0.023	0.030	0.025	0.026	0.024	0.024	0.023	0.023	0.023
Cd	0.001	-0.001	0.001	-0.001	0.000	-0.001	-0.001	-0.001	-0.001	-0.001
Hg	0.000	0.000	0.000	0.000	0.002	0.002	0.002	0.002	0.002	0.002
Cu	0.13	-0.18	0.22	-0.09	0.09	-0.16	-0.16	-0.23	-0.27	-0.29
Cr	0.055	0.053	0.057	0.055	0.056	0.054	0.054	0.053	0.053	0.053
Ni	0.01	-0.09	0.01	-0.09	-0.01	-0.09	-0.09	-0.11	-0.13	-0.13
Zn	0.37	-0.86	0.50	-0.76	0.31	-0.68	-0.67	-0.94	-1.08	-1.16

[1] Net soil contamination sometimes not the exact difference due to rounding-off errors