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Abstract

In their fourth assessment report the IPCC stated that it is very likely that a causal connection exists between human activity, greenhouse gas emissions and global warming (IPCC, 2007a). Reducing the level of greenhouse gas emissions is on the policy agenda in many countries, including Norway. Human settlements draw on resources and cause emissions in many different ways, for example through direct energy use, transportation, water use, wastewater treatment and solid waste handling. The level of impact is influenced by the lifestyle of the inhabitants. Other countries have started both research on and the building of carbon neutral settlements in order to reduce their national carbon footprints. Great Britain has ambitious goals to become a carbon neutral society, and low carbon settlements are seen as one measure to achieve this; the low carbon BedZED project near London is completed and several other low carbon settlements are planned (Holt, 2008). In Copenhagen the Carlsberg Corporation is planning to build a carbon neutral neighbourhood in the old Carlsberg factory area, and in Sweden “Hammarbymodellen”, an eco-cycle model, provided the foundation for ambitious environmental objectives during the planning of Hammarby Sjöstad, a settlement in Stockholm with 15,000 inhabitants (Finnson, 2006). In 2007 an initiative was started with the aim of creating the first Norwegian carbon neutral settlement at Brøset in Trondheim. To contribute to the planning process of this ambitious settlement, the Norwegian Research Council founded the project, “Towards carbon neutral settlements - process, concept development and implementation”. This thesis consists of environmental assessments of various potential infrastructure solutions for the waste, water and wastewater systems at Brøset.

Today’s urban infrastructure is the result of centuries of work for healthy city environments. Most developed countries have efficient and highly developed infrastructure for solid waste, water and wastewater, and this is the case in Trondheim. The carbon neutral settlement planned for Brøset will have approximately 1600 dwellings, and will be situated four kilometres from the city centre. The site itself is mainly a greenfield area, but the surrounding built environment is suburban. Trondheim has well-functioning waste and water infrastructure, based upon services provided by centralised facilities. These systems draw on resources in the form of energy and material, process resources as nutrients and energy, and result in direct and indirect emissions from treatment and disposal. We were interested in estimating the role of infrastructure in the overall impact from the Brøset settlement, and to assess whether a new settlement with carbon neutral ambitions should adjust to the conventional infrastructure or implement alternative centralised or decentralised solutions.

The assessment of the infrastructure systems involved in proposed new settlements with ambitious environmental objectives requires specific methods. This thesis will demonstrate how to structure an environmental model analysis of water, wastewater and waste infrastructures for urban settlements. Carbon neutrality is the main objective for the settlement, but other environmental impacts are included in the calculations. This was done in order to avoid, or at least be aware of, possible problem shifting and environmental trade-offs.

To assess the importance of the supporting infrastructure, and to compare alternative systems for the infrastructure, we followed five steps. First, life cycle assessment (LCA) was used to assess the impact of the existing water and wastewater systems in Trondheim and a “business-as-usual” household waste system. Second, the literature was searched for state-of-the art research on innovative new solutions for water, wastewater and solid waste systems that were suitable for the situation at Brøset. Third we used the information from the literature and a parallel commissioning process, and data from Trondheim municipality and from the assessments of the systems in the first step, to build alternative scenarios. Fourth, we used LCA to compare alternative technical solutions for the systems. Finally we interpreted the results of the assessments.

For the waste management system in Trondheim the total impacts in most impact categories were found to be negative, representing a saving in impact due to substitution. Substitution is in this case the replacement of virgin production of materials and energy, and the impacts from this production. Mixed waste is used to provide heating for the district heating system in Trondheim, and thereby replaces other energy sources. Recyclables are used to substitute materials and energy. When measures such as increased recycling and introduction of food waste sorting and biogas production were assessed, the results showed only small differences among the scenarios, although some benefits from increased source-separation of paper and metal were found. The settlement should therefore be connected to the existing waste management system of the city, and not resort to decentralised waste treatment or recovery methods.

For the water and wastewater system the life cycle global warming impact per person in the city of Trondheim was found to be less than 1 % of the annual total per-capita impact. Around 54% of this was attributed to the operation and discharges from the wastewater treatment plants. The alternative systems for Brøset were few, due to the low total impact. Some improvement in impact could be found when water consumption was reduced and stormwater handled locally, but the gains were small. Source-separation of wastewater and treatment of the greywater in constructed wetland was found to have higher impact than connecting to the conventional system in some impact categories, including global warming.

Although there is an extensive body of research available in the waste, water and wastewater fields, we found two important areas that have received little attention in the literature. These were analysis in waste management assessments of uncertainty due to differences in waste composition, and the issue of waste prevention. A conceptual study of the consequences of uncertainty in waste composition was performed, and other sources of uncertainty in the assessments carried out for this thesis were discussed. In order to account for waste prevention in environmental assessments we have developed a model that includes the impact from the production of goods in the assessment. This was performed using a hybrid-LCA model, in which the upstream impact was modelled with environmentally extended consumption-based input–output analysis and the downstream waste system was modelled with LCA. The importance of upstream impact became evident, but also the importance of including rebound effects in the calculations.

A thesis is written in a given time period and there are always several issues that could be followed up with more research. The waste prevention model is in an early stage of development. The model should be developed further as waste prevention as a research field deserves greater attention, in terms of both estimating the effects of and developing successful measures for actually achieving waste prevention.

There are two overall take-home messages from this work. The first is related to the availability and usefulness of existing methods to evaluate environmental impacts from an urban development project in its early phase of planning. Here we conclude that the combination of system analysis and scenario building were helpful in the early stage planning phase for assessing the role of the infrastructure, for including several environmental impact categories, and for comparative assessments of alternative solutions. The second take-home message is related to what kind of strategies and solutions for technologies and management in the waste, water and wastewater subsystems that are to be recommended, in a case such as Brøset in Trondheim. Here we conclude that Brøset should connect to the existing infrastructure systems, but that local stormwater treatment and measures for waste prevention and water saving should be integrated in further planning.

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My English writing is not very good and I would therefore like to thank Venkatesh and Simon Loveland for proof reading my writings. The life cycle assessments performed in the thesis would not be possible without a lot of data input, I would therefore like to thank all the people in the municipality in Trondheim who provided me with valuable information for my assessments.

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List of papers

Paper 1 Slagstad, H. and Brattebø, H. (2012). LCA for household waste management when planning a new urban settlement. *Waste Management* 32, 1482-1490.

Paper 2 Slagstad, H. and Brattebø, H. (2013). Influence of assumptions about household waste composition in waste management LCAs. *Waste Management* 33, 212-219.

Paper 3 Slagstad, H., Lébre, E. and Brattebø, H. Using IO-LCA to explore how household waste prevention influences economy-wide GHG emissions. Submitted to Waste Management April 2013.

Paper 4 Slagstad, H. and Brattebø, H. Life cycle assessment of the water and wastewater system in Trondheim, Norway – A case study. Accepted for publication in Urban Water Journal.

Paper 5 Slagstad, H. and Brattebø, H. Use of LCA to evaluate solutions for water and waste infrastructure in the early planning phase of carbon-neutral urban settlements. Accepted for publication in Smart and Sustainable Built Environment. Available online.

Paper 6 Slagstad, H. and Brattebø, H. Environmental impact of water, wastewater and waste infrastructure. Draft to a book chapter in a book about the planning process of Brøset. Submitted to review.

The author’s contribution

The papers are co-authored. The author of the thesis has performed the following work for the papers:

Paper 1 Data collection, modelling, analysis and writing.

Paper 2 Data collection, modelling, analysis and writing.

Paper 3 Data collection on the waste system. Contributed to the modelling of the waste system. Writing.

Paper 4 Data collection, modelling, analysis and writing.

Paper 5 Data collection, modelling, analysis and writing.

Paper 6 Data collection, modelling, analysis and writing.

Chapter 1

Introduction

Background and motivation

The relationship between greenhouse gas emissions, rising temperatures, and changes in the global climate – and the potentially significant consequences of the latter – described by the Intergovernmental Panel on Climate Change (IPCC) has led to greenhouse gas emissions becoming one of the main concerns for sustainable development (IPCC, 2007a). A changing climate will have consequences for people's ability to grow food and survive in many places in the world, and there is also concern that climate change may become self-enforcing, reaching such a level that will make it impossible to prevent further change. Sustainable development was in 1987 defined as '*developments that meet the need of the present without compromising the ability of future generations to meet their own needs'* (World Commission on Environment and Development, 1987). The IPCC stresses the importance of slowing the rate of increase and thereafter reducing the level of greenhouse gas emissions to the atmosphere in order to make it possible for ecosystems to adapt to a changing climate, secure global food production and enable sustainable economic development. In 2007 a white paper established sustainability as a fundamental principle for all development in Norway (Ministry of finance, 2007). Key strategies for the achievement of sustainable development that are highlighted in that paper are fair distribution, international solidarity, the precautionary approach to environmental impact, the polluter pays principle and joint effort. Sustainable development comprises the interaction between its environmental, economic and social aspects, and all aspects have to be fulfilled in order for sustainability to be achieved.

Under the Kyoto Protocol Norway committed to limit its annual emissions in the period 2008–2012 to a level no more than one per cent higher than in 1990. The awareness of climate change and greenhouse gas emissions has increased since the Kyoto negotiations. A Norwegian white paper on climate change published in 2012 established carbon neutrality by 2050 as Norway's long-term goal, with 2030 as an alternative if other countries also commit to that timescale (Ministry of environment, 2012). The short-term goal is a 30 % reduction in emissions by 2020 compared to the emission level in 1990. The targets for Norway are based on two-thirds of the emissions cuts being achieved through national measures, while climate quotas can be used for the remaining third. This is in accordance with the United Nation's definition of carbon neutrality at a country, city or company level (UNEP/GRID-Arendal, 2008).

There is no doubt that reduction of greenhouse gas emissions is a genuine goal for both the UN and the Norwegian government, and that this is based on the desire to achieve a sustainable future. However, reducing emissions to zero is challenging, and involves complex economical, ecological and social systems. According to Hertwich and Peters (2009) the level of per capita greenhouse gas emissions varies widely between countries, ranging from 1 tonne CO_2-equivalet per capita annually in some African countries to approximately 30 tonnes CO_2-equivalent per capita in the USA and Luxembourg. Norway has an average impact of 14.9 tonnes CO_2-equivalent per capita per year.

Half of the world's population lives in cities, and 80 % of the world's greenhouse gas emissions are assumed to be connected to the activities of urban residents and their affluent lifestyles (Hoornweg et al., 2011). Hertwich and Peters (2009) found 72 % of global greenhouse gas emissions to be related to household consumption, with nutrition and shelter being the two most important consumption categories. In Norway, however, mobility and service were on average the two most important categories. According to BioRegional and CABE (2008) the CO_2-emissions of an average UK resident can be divided into eight categories. Domestic energy use and transportation are the two largest categories (23 % each), followed by consumer goods (13 %), business (10 %), housing, food and government (8 % each), and finally capital assets (7 %). There are, however, large differences in average per capita emissions depending on whether calculations are performed at a country, city or neighbourhood scale. This is because the level of impact is influenced by factors such as income, housing patterns, availability of public transport, extent of industrial development, energy sources in use and the local climate. VandeWeghe and Kennedy (2007) showed how the impact of an inhabitant of Toronto can vary from 1.3 tonne CO_2-equivalent per capita to 13.0 tonnes CO_2-equivalent per capita depending on where he or she lives in the city.

The future development of cities will involve the improvement of both the systems and buildings we have in place already and those which we are adding to the urban environment. Farreny et al. (2011a) define a sustainable settlement as an "urban settlement that is adapted to the local environmental characteristics and makes efficient use of resources especially local, or else regional, minimises its emissions, and shows an increase in quality of life (including aspects of health, education, and welfare) without compromising the carrying capacity of the natural environment, so it can better fit within the capacities of the local, regional, and global ecosystem". Carbon neutrality is the objective of a new settlement planned at Brøset in Trondheim, Norway. The 35-hectare Brøset area is a greenfield site within the city about four kilometres from the centre. Approximately 1600 dwellings are planned, and carbon neutrality has been the main objective of the development from the beginning.

Complete carbon neutrality, however, was found to be too challenging without buying climate quotas, and the goal is now therefore to achieve an impact below 3 tonnes CO_2-equivalent per person annually, compared to the average Norwegian impact of 14.9 tonnes CO_2-equivalent per capita per year (Hertwich and Peters, 2009). This remains a challenging goal when the difficulty involved in influencing some of the emissions is taken into consideration, for example emissions from food and goods produced abroad.

While research into and the building of carbon neutral settlements have begun in other countries, experience of this type of project is limited in Norway. Norway is a distinctive country, with its variety in climate, relatively small cities, comparatively high per capita environmental impact, the use of electricity as the main heating source in buildings, very little landfilled waste and abundance of water resources. In order to ensure the success of the planning phase of the Brøset settlement, the "Towards carbon neutral settlements - process, concept development and implementation" project, funded by the Norwegian Research Council, was initiated in 2009. The project focuses on how factors such as transportation, housing, energy systems, infrastructure and lifestyle affect carbon neutral settlements. Several articles and reports have been published so far; Gransmo (2012) discussed municipal planning of a sustainable neighbourhood, focusing on action research and stakeholder dialogue; Thomsen (2011) reflected on the opportunities of urban planning to promote non-vehicular transportation, and Solli et al. (2010) described the causes of emissions from residential areas, identified focus areas and estimated the potential for reductions in impact.

It is our contention that standard design processes in Norway, when not completely lacking in environmental objectives, often focus on low-energy solutions for heating. This is of course an important area of interest, CO_2-emissions related to energy used for heating being very significant. There is, however, less research into and knowledge of the relationship between supporting infrastructures, and the importance of these infrastructures for the consumption of resources and generation of emissions. In order to build carbon neutral settlements we need to bridge the gap between promising infrastructure research and its actual utility in urban-development projects. This thesis covers the infrastructures of urban water, wastewater and household waste management. In order to be able to both understand the importance of new settlements' supporting infrastructure and compare different solutions during the early stages of the planning phase we need methodologies for the systematic assessment of the systems involved. Industrial ecology is an interdisciplinary field defined by White (1994)as *"the study of the flows of materials and energy in industrial and consumer activities, of the effect of these flows on the environment, and of the influence of economic, political, regulatory and social factors on the flow, use and* *transformation of resources".* Industrial ecology methods include life cycle assessment (LCA), material flow analysis (MFA) and input–output analysis (IOA). LCA has been used within the fields of waste, water and wastewater research for many years. In the waste field LCA has been used to assess the waste management systems of countries and cities (Cherubini et al., 2009; Eriksson et al., 2005; Larsen et al., 2010; Raadal et al., 2009), waste fractions (Astrup et al., 2009; Larsen et al., 2009a; Merrild et al., 2009) and particular elements of the waste management system (Eisted et al., 2009; Rives et al., 2010). Water and wastewater systems have also been assessed at different scales, ranging from entire water and wastewater systems (Hofman et al., 2011; Lassaux et al., 2007; Lundie et al., 2004; Venkatesh and Brattebo, 2011) to detailed LCA of processes (Anand and Apul, 2011; Fuchs et al., 2011). There are fewer examples of the use of MFA and IOA on these systems. However, some examples can be found; Eckelman and Chertow (2009) used MFA to demonstrate long-term waste management solutions in Oahu, Hawaii, and Nakamura and Kondo (2002) developed an input–output model for assessments of waste management systems.

In the literature there is a rising concern that today's infrastructure systems are unsustainable. Guest et al. (2009) and Larsen et al. (2009b) asked for a paradigm shift in wastewater handling. Astrup (2011) stressed the importance of seeing waste as a resource, and Agudelo-Vera et al. (2011) claimed that bringing resource management into urban planning is one of the most important steps towards sustainable urban planning. Meijer et al. (2011) described the importance of moving from end-of-pipe solutions to a cyclic metabolism at a city level. Waste-to-energy plants and the use of sludge as fertiliser are examples of how resources can be utilised several times within a city. The self-sufficiency cyclic metabolism can also be achieved within a settlement, with decentralised solutions for wastewater treatment, water recycling and local food waste handling.

The project team responsible for the planning of the Hammarby Sjöstad settlement in Stockholm, Sweden developed the Hammarby model (Finnson, 2006). This large, new sustainable settlement combined conventional systems with some local solutions, and the model shows how the new settlement both contributed to and utilised the resources in the surrounding city. Crewe and Forsyth (2011) describe Hammarby Sjöstad as an ecocity project, a project characterised by a compact approach, with density, energy recovery and use of reclaimed land as focus areas. In contrast, they explain, ecoburbs are leafy and natural looking, and include decentralised systems and food production. One example of an ecoburb is the rural settlement of Flintenbreite in Germany (GTZ Ecosan project, 2005). This 5.6-hectare ecological settlement west of Lübeck was part of EXPO 2000 Hanover, and has 117 accommodation units. Flintenbreite is disconnected from the sewer system and uses blackwater and food waste to produce energy on site. Greywater is treated in constructed wetlands.

While decentralised systems are usually applied to rural settlements, there are examples of how such systems can be implemented in urban areas. The Klosterenga project in Oslo provides an example of urban greywater treatment in a cold climate using constructed wetlands (Jenssen et al., 2005). The greywater from 33 apartments has been treated in the neighbourhood courtyard since 2000. The greywater is collected in a septic tank and pre-treated in a biological filter prior to insertion in the subsurface horizontal-flow constructed wetland. This concept was first tested in Norway in 1991, with good performance results and little maintenance required. One square metre of wetland is required per inhabitant, and the treated wastewater can be discharged into local streams. These alternative, local systems have, however, trade-offs economically, technically and in the use of energy. Remy (2008) found that if the conventional system is energetically optimised a source-separation system does not necessarily have less environmental impact than a conventional system. If carbon neutrality is the objective this has to be kept in mind.

The Beddington Zero Energy Development (BedZED) in London, England has received a lot of publicity for its achievements as a low carbon settlement. However, there have been difficulties with the water and wastewater systems. The settlement, built in 2002, contains 82 dwellings and some commercial space. The settlement features rainwater harvesting, local food production and local wastewater treatment, which originally used a low-energy treatment system. The complexity of the Living Machine wastewater system proved challenging to operate, however, and a membrane bioreactor (MBR) for local treatment of the wastewater was installed instead. The membrane bioreactor turned out to be a more costly and energy intensive system than the conventional systems available (Shirley-Smith and Butler, 2008). There were also problems with the rainwater harvesting system and the use of non-potable water for flushing the toilets. In spite of all these problems, BedZED is a very important pilot project because of the lessons that can be learned for the installation of local water and wastewater systems in urban settlements.

There are several ways of managing stormwater in a city. One method is to collect it together with sewage and transport it to a centralised wastewater treatment plant. Another is to collect and transport it separately and release it directly to a surface water body. A third option is to manage the stormwater locally. Separate stormwater collection systems have for many years been the preferred solution. Such systems reduce the amount of water going to the wastewater treatment plant. However, older, combined sewers still account for 50 % of the system in cities such as Trondheim. The drawback with combined sewers, in addition to the extra water going to the wastewater treatment plant, is the overflow system. Overflow during heavy rainfall is a point source of nutrient rich wastewater escaping to the environment. Local treatment of stormwater is gaining increased interest in Norway and many other countries. The

Hammarby Sjöstad and Augustenborg projects in Sweden are examples of how measures such as the separation of polluted and unpolluted stormwater, infiltration, and the use of vegetation and green roofs can be used to reduce the stormwater load on the wastewater system. However, the winter conditions in Trondheim, with several months of snow and ice, are challenging for the infiltration process.

There are fewer examples of local waste management solutions, except for home composting of food waste. Waste management systems have been shown to be responsible for approximately 2 % of the total greenhouse gas emissions globally, originating primarily from organic waste in landfill sites and the incineration of wastes with a fossil origin (McKinsey and Company, 2009). Well-developed household waste management systems with limited use of landfill are shown in several studies to produce savings in global warming impact due to substitution of energy and materials (Christensen et al., 2009; Gentil et al., 2009; Raadal et al., 2009). Based on the impact of different treatment options, the waste hierarchy – reduction, re-use, recycling, incineration, landfilling – has been validated as a rule of thumb by Finnveden et al. (2005). Norwegian waste policy has been guided by the waste hierarchy since the early 1990s, and landfilling of organic waste was banned in Norway in 2009. In 2010 50 % of the household waste in Norway was incinerated, 42 % was sent for recovery, 6 % was landfilled and 2 % received other treatment (SSB, 2011).

Waste prevention is defined as the reduction and reuse of waste, the two measures at the top of the waste hierarchy. Wilson et al. (2010) called for more attention be paid to the waste prevention issue, due to the fact that in many countries (such as Norway) high recycling rates and the use of incinerators with energy recovery are already well-established. A Norwegian Official report on waste prevention was published in 2002 (Ministry of environment, 2002). While Norway is already close to its goal of 75 % recycling or energy recovery, the actual amount of household waste has been steadily increasing for many years, and waste prevention has not been successfully implemented in practice. According to Sharp et al. (2010) the potential for waste prevention is assumed to be around 0.5 to 1 kg/household/week, with the greatest potential found in the food, garden and bulk waste fractions. However, estimating the environmental effects of waste prevention is challenging and the field is not well researched, according to the literature.

Farreny et al. (2011a) describe how the inclusion of sustainability criteria during the early stages of the design and planning of urban systems is the best strategy for environmental protection. However, the implementation of criteria at an early stage relies on knowledge of what the most important criteria actually are. There are examples of how settlements with low-impact ambitions have introduced solutions that are more complicated and expensive and have higher environmental impacts than would have been the case with conventional systems. Other settlements do not utilise the solutions available and appropriate in their local context. Having objectives that are too narrow or using sustainable indicators based on perception rather than knowledge of the least-impact solutions, without considering the context of the new settlement, can thereby lead to poor results.

When examining the environmental qualities of possible new solutions at the Brøset project we have included several environmental impact categories in addition to global warming (which covers the issue of carbon neutrality). This was seen as important in order to both compare the importance of greenhouse gas emissions with other environmental impacts, and be aware of potential environmental trade-offs when comparing alternative technical solutions. We have not included economic and social aspects of sustainability in this thesis. These are, however, important, and discussions around the broader definition of sustainability have been included in the planning of the Brøset project.

Research questions

There is substantial research into the environmental performance and carbon emissions of the technologies involved in the infrastructure systems that provide the focus for this thesis. However, there are few examples of a holistic approach, where all these systems are studied together, in combination with physical planning, and in the context of carbon neutral settlements. In addition there is little research on the performance of the existing waste, water and wastewater systems in Norway. In order to build carbon neutral settlements we need to bridge the gap between promising infrastructure research and its utility in specific urban-development projects. This thesis will contribute to the discussion surrounding the physical planning of sustainable urban settlements and cities by focusing on the environmental assessment methods used during the early stages of the planning phase. Existing infrastructure systems are used as starting points, with international successful case studies and promising technologies used as inspiration for new solutions. With better knowledge of critical factors of infrastructure design and operation, and of promising technological solutions, we may contribute to the development of carbon neutral settlements and improve the early stages of the planning process of such settlements.

The research questions are:

1. What are the promising concepts for water, wastewater and waste infrastructure design and operation that could contribute to achieving a carbon neutral settlement at Brøset in Trondheim?

2. How do such concepts contribute to improvements in resource consumption, emissions and life cycle environmental impacts, particularly with regard to greenhouse gas emissions?

3. How can the urban settlement planning process benefit from identification and assessment of such concepts?

The research questions will be answered by the papers included in this thesis and by the thesis text itself. A detailed summary related to each question is found in *Contribution of papers* section included in Chapter 3.

Chapter 2

Research methods

System analysis was used to assess and compare different infrastructure systems in order to provide decision support to the planning phase of the new settlement at Brøset. Several methods were applied, including literature review, scenario building, life cycle assessment (LCA) and hybrid-LCA (a combination of LCA and input–output analysis (IOA)). Uncertainty analysis is important when assessing complex systems. Uncertainty is discussed in relation to each assessment, below; in addition a conceptual study of the consequences of uncertainty in waste composition was performed. LCA and hybrid-LCA are described in general below, before a description of the methods used to carry out environmental analysis of the infrastructure in Trondheim and for the project at Brøset specifically.

Life cycle assessment

Life cycle assessment is a methodology covered by ISO 14040 and 14044 (ISO, 2006a, b). It was originally used for cradle-to-grave assessment, where the environmental impact of all inputs from and outputs to the environment in a production chain is estimated. An increasing number of advanced systems have been assessed since then. According to the ISO standard there are four stages in an LCA; (1) goal and scope definition, (2) life cycle inventory (LCI), (3) life cycle impact assessment (LCIA) and (4) interpretation (ISO, 2006a). The first step establishes the goal and the context of the assessment. System boundaries, in terms of both time and the processes to be included, are important for the outcome of the LCA. The second step, LCIA, quantifies the inputs to and outputs from every process in the defined system. For this thesis, this step consisted of gathering data to describe the amount of waste, the volume of water and wastewater, the direct emissions, the energy used in the different processes and so on. This step requires detailed information on the different processes and parameters involved. The third step uses characterisation factors to aggregate the emissions into impact categories. As an example, global warming is an impact category measured in CO_2-equivalent, with CO_2 having a characterisation factor of 1, while methane, CH_4, has a characterisation factor of 25 (IPCC, 2007b). Methane emissions are thereby multiplied by 25 in order to convert them into CO_2-equivalent.

Normalisation of the emissions values reveals the relative importance of the results, and makes it easier to compare across impact categories. Average impact per person in Europe (or the world) can be used as a normalised value. The last step of an LCA is to interpret the results. In this step evaluation of completeness, sensitivity and consistency is important. LCA is an iterative process, as part of which we have to go back and forth between the steps in order to improve the results. The results can be weighted and combined into a single score, but this is optional and was not carried out for this thesis.

LCA does not take only direct emissions into account, but also includes impacts resulting from the production and transportation of resources, construction and maintenance of buildings and infrastructure, end-of life management, and so on. Ekvall et al. (2007) evaluated the use of LCA in waste management research and explained the importance of the indirect environmental impacts for the total impact of a system. Using LCA provides great opportunities for the environmental evaluation of the systems under study, but it also has limitations. Gentil et al. (2010) reviewed the importance of technical assumptions in models for waste management and found that the functional unit, system boundaries, waste composition and energy modelling all have a significant impact on the results. Consistency and transparency are therefore important when performing LCA, and an LCA handbook is available from the Joint Research Centre of the European Commission to help with achieving these aims (EU JRC, 2010).

Although LCA can be data intensive and time consuming, there are many tools available and two of these were used in the research for this thesis. Easewaste is a designated LCA tool for waste systems, which allows for waste to be followed from collection to final destination (Kirkeby et al., 2006). The impact method was EDIP 97. The environmental impacts were normalised according to EDIP97 values of global or EU-15 annual environmental impacts for one person, and the results are given in person equivalents (PE). Easewaste was updated in 2012 and both the old and the new version were used for this thesis. The alternative waste management scenarios (Paper 1) were assessed with Easewaste 2008, using the normalisation values given in Christensen et al. (2007). The waste composition uncertainty analysis and the waste prevention assessment (Paper 2 and 3) were performed using Easewaste 2012. The normalisation values for the last two assessments were provided by Laurent et al. (2011).

For the assessment of the water and wastewater systems Simapro (Pré Consultants, 2011), a more general LCA tool, was used in combination with Excel. Simapro includes multiple databases, of which Ecoinvent was used in these calculations. The impact assessment method applied was ReCiPe midpoint (H) v1.06, July 2011, and the impacts were normalised against average annual emissions per person in Europe, and presented in person equivalents. For N_2O emissions the IPCC characterisation factor from 2007 was used (298 kg CO_2-equivalent per kg N_20) (IPCC, 2007b). We excluded the impact categories dealing with toxicity from the assessments of the water and wastewater system. This was due to a lack of data on toxic elements in stormwater overflows, wastewater effluents and sewage sludge.

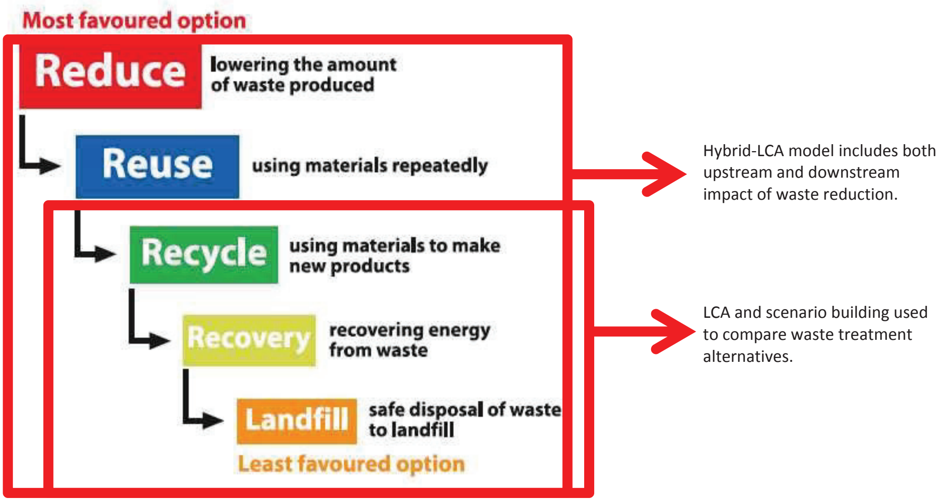
One important LCA parameter that is often discussed is energy. The discussion concerns the electricity production, and in particular whether to use average or marginal data to describe this. While an average electricity mix can be, as in the Norwegian case, fairly clean, marginal electricity production can be based on, for example, coal. A change in the electricity mix can therefore alter the results of an assessment. The ILCD handbook recommends that marginal electricity data should be used only when describing systems that have a significant impact on the energy use of a country, for example when performing consequential LCA on policy measures (EU JRC, 2010). Changes in the infrastructure systems assessed in this thesis would not affect the Norwegian energy system; average data was therefore used. In accordance with practice in the rest of the Brøset project the Nordic electricity mix was used for the assessments due to the strong connection between the Norwegian and the Nordic electricity markets.

Input-output analysis and LCA

Input–output analysis (IOA) is, in contrast to LCA, characterised by a top-down approach, where we relate environmental impacts to monetary rather than physical flows. IOA covers the entire economy of a nation and features a high level of aggregation. This method is therefore not suitable for detailed studies at an industry level. The different characteristics of LCA and IOA in terms of aggregation level and system extent can be utilised in different hybrid methods. LCA can be used for calculating the detailed foreground system while IOA can be used for the more aggregated background system. In this way the advantages of each method can be utilised. In the case of the waste management system we combined the two methods by applying an environmentally extended input-output consumption-based model to the upstream emissions related to the production of goods, while LCA was used on the waste management system itself. This model is discussed further later in this chapter.

The waste system

The waste hierarchy has been validated as a rule of thumb (Finnveden et al., 2005) and provided the starting point for the assessments related to the waste field in this thesis (Figure 1). The waste system in Trondheim is mainly based on incineration and recycling. We wanted to assess the importance of moving the waste up the waste hierarchy. We first used LCA and scenario building to assess scenarios describing biological treatment of food waste and increased recycling of paper, plastic, glass and metal. The waste system was modelled using a no-burden approach, in which waste enters the waste system without consideration being taken of any upstream emissions from the production and use phases. This is a common approach in LCA of waste systems (Bjarnadóttir et al., 2002). However, the no-burden approach fails to deal adequately with waste prevention, which, with its objectives of reducing and reusing, is at the top of the waste hierarchy. A hybrid-LCA model was therefore developed to deal with the change in environmental impacts related to waste prevention. The hybrid-LCA model will be explained later in this chapter; we will first concentrate on the assessment of various potential technical solutions available in the lower part of the waste hierarchy.



The waste hierarchy is sourced from http://wasteawarebusiness.wordpress.com/ and modified.

Figure 1. Waste research related to the waste hierarchy.

The functional unit was defined as 'the collection, transport and treatment, during one year, of the waste streams of mixed waste, paper, plastic, glass and metals from 1500 new households (3315 persons) at Brøset in Trondheim, Norway'. Source separated hazardous waste, EE-waste, garden waste, wood, and bulky waste were left out of the calculations. These waste streams would not in this case have been affected by changes to the waste system induced by introducing alternative systems.

There are three main categories of options that are important for the results of a waste management LCA: (1) system boundaries, (2) waste composition and sorting efficiencies, and (3) technical solutions including energy choices. The waste system modelled for the business-as-usual case at Brøset is shown in Figure 2.

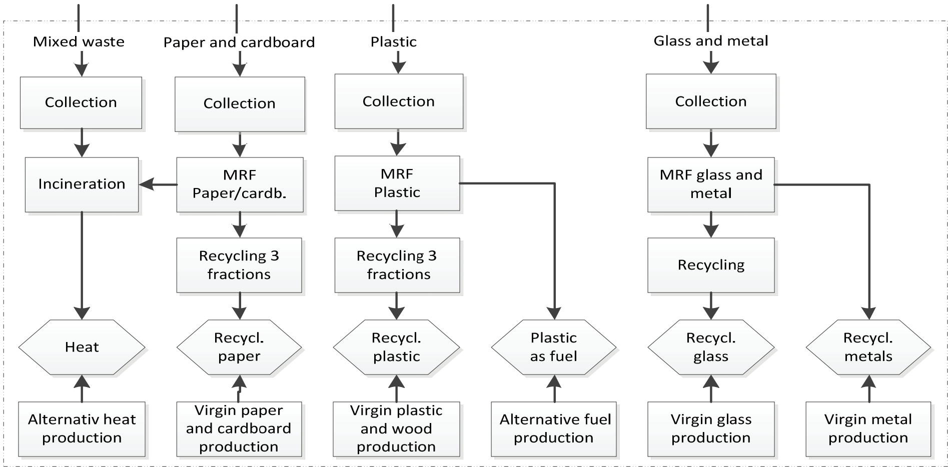


Figure 2. The business-as-usual waste system modeled for Brøset.

System boundaries define which processes are included in an assessment. Substitution of materials and energy were included in the modelled waste system, this being a common approach in assessments of waste management systems (Bernstad et al., 2011; Christensen et al., 2009; Gentil et al., 2009; Raadal et al., 2009). Substitution can in well-developed waste management systems lead to beneficial impacts in many impact categories when the substituted energy or material is more emissions intensive than the recycling process. There are, however, some challenges regarding where to set the boundaries when substitution is included. One example is paper recycling. The recyclable paper fraction usually replaces virgin paper production. Virgin paper is produced from wood that could otherwise either stay in the forest or be used as an energy source. Christensen et al. (2009) include this alternative use of wood in their assessment of 40 different waste systems. The problem with including alternative uses of raw materials is the complexity of the relationship between the different processes. Alternative use of wood was not included in any of the assessments in this thesis, in line with comparable assessments from Raadal et al. (2009), Gentil et al. (2009) and Bernstad et al. (2011).

Waste composition and sorting efficiencies for Brøset was estimated based on the literature and available data on waste composition and amounts of waste sorted (Astrup et al., 2009; Damgaard et al., 2009; Larsen et al., 2009a; Merrild et al., 2009; Raadal et al., 2009). There was uncertainty in the estimated waste composition, and the consequences of this uncertainty were tested in a separate analysis (see Paper 2).

Turning to the technical solutions in use in Trondheim, we knew that a large fraction of the town's waste is incinerated with heat recovery. Up-to-date information about emissions from the incinerator was available. In addition we knew which energy sources were substituted through utilisation of the heat from the incinerator in Trondheim's district heating system (Brattebo and Reenaas, 2012). While the mixed waste was sent directly for incineration, the three source-separated fractions – paper, plastic and glass/metals – were sent to Material Recovery Facilities (MRFs). For each MRF waste transfer coefficients had to be decided. The impact from the recycling process was determined by the amount of waste recycled, the impact from the recycling process and the impact from the substituted material or energy. Recyclables are today traded in markets and it can therefore be challenging to model the destination of the recyclables, and there can be large differences between technologies (Merrild et al., 2008).

The five scenarios assessed in the waste system were:

Scenario 1: Business-as-usual
Scenario 2: Source-separation of food waste, a centralised biogas plant, upgrading of biogas to fuel, the other fractions as in the business-as-usual scenario.
Scenario 3: Source-separation of food waste, local biogas plant, biogas used in a combined heat and power plant, the other fractions as in the business-as-usual scenario.
Scenario 4: Increased recycling, 90% source-separation of paper, glass and metals, 70 % source-separation of plastic.
Scenario 5: Combination of scenario 2 and 4.

Uncertainty

Uncertainty has to be accounted for when performing LCA. In the comparison of different waste management solutions for Brøset, the uncertainties in the technological and energy parameters were tested by applying alternative parameters. The electricity mix was changed to, firstly, the Norwegian electricity mix and, secondly, the European electricity mix, which have lower and higher environmental impacts, respectively, than the Nordic electricity mix used in the original assessment. The technological parameters were tested by modelling the paper and metal recycling technologies with more generic technologies. Uncertainty in waste composition and sorting efficiencies were not considered in the Brøset assessment. There is, however, a lack of literature covering the consequences of uncertainty in waste composition and a conceptual study was therefore conducted. The system was in this case modelled to represent a typical Norwegian city, including incineration with heat recovery and source-separation of paper, plastic, glass and metals. However, the system was simpler than that envisaged for Brøset, with fewer waste fractions and recycling routes. Treatment of residues from the incinerator was also excluded. We used two different methods for deciding the composition and amount of waste entering the different treatment options (Figure 3). Case 1 had a constant sorting efficiency. By changing the waste composition we obtained variation in the amounts of waste both recycled and incinerated, and variation in the waste composition of the incinerated waste. In Case 2 the amounts of waste recycled and incinerated were constant. Therefore variation occurred in sorting efficiency and in the waste composition of incinerated residual waste.

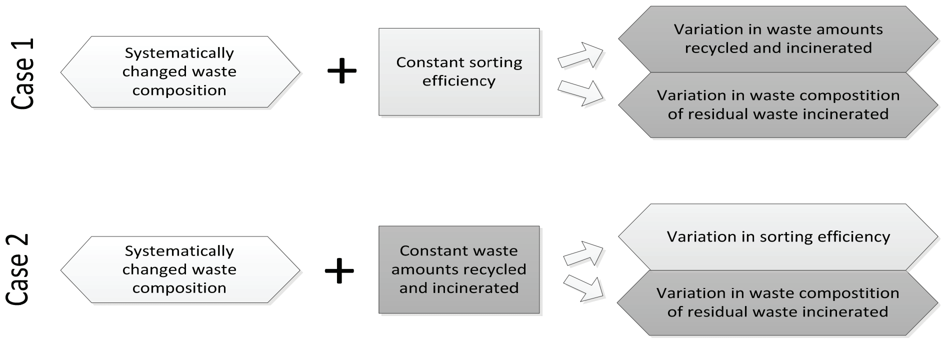


Figure 3. Two methods estimating the consequences of changes in the waste composition. Case 1 is for new systems, or systems with changes in technology, while Case 2 is for existing systems.

More detailed information on how the system was modelled and the waste composition systematically altered can be found in Paper 2.

Waste prevention

Waste prevention was modelled using a hybrid-LCA method, combining IOA for upstream impact with LCA for downstream impact (Figure 4). For the upstream impact the environmentally extended input–output database, EXIOBASE (EXIOPOL, 2012), was used in combination with a consumer expenditure survey to model the impact from consumption, while the Easewaste LCA waste tool was used for the waste system. In this way we were able to include the rebound effect, where money saved by reduced consumption in one category is spent in other consumption categories. The environmental impact of these other consumption categories is often important for the total impact of the system. The model is explained in more detail in Paper 3.

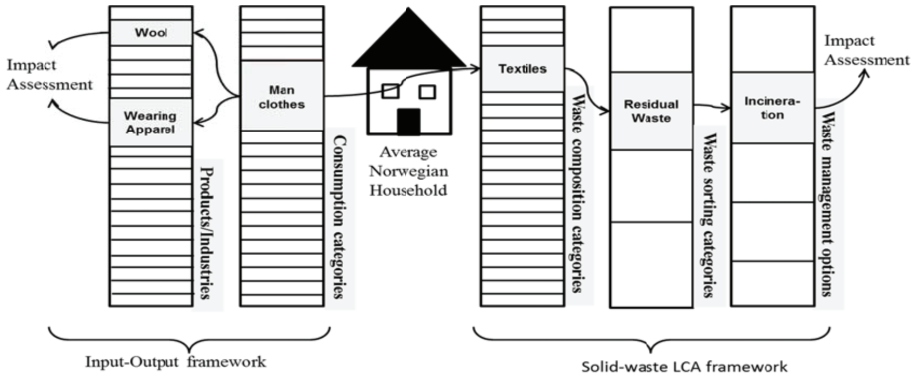


Figure 4. Hybrid-LCA for waste prevention calculations (Lébre, 2012).

The water and wastewater system

Hamouda et al. (2009) reviewed decision support systems for the selection and design of water and wastewater treatment processes, and found that technical considerations still dominated the available tools. Although some environmental assessments were available that estimated the total environmental impact of water and wastewater systems and described the important parameters and impact categories in such systems, more research is needed. This is why the entire water and wastewater system of Trondheim was analysed for this thesis in a separate study not directly focusing on the Brøset project. The system is shown in Figure 5 and is also explained in more detail in Paper 4. There are two major wastewater treatment plants and one water treatment plant in the city. We had access to detailed information on these processes, including energy use, input of materials in the different treatment facilities, biogas production, the level of phosphorous before and after wastewater treatment, and the use of sludge. We also had access to data on the pipe system in Trondheim, and the number of pumps and water storages facilities and related electricity use.

There are no nitrification/denitrification processes in the wastewater treatment plants and the nitrogen content of the wastewater is not measured. Nitrous oxide (N_2O) is a potent greenhouse gas with a characterisation factor 298 times that of CO_2 (IPCC, 2007b). In the absence of a nitrification/denitrification process, the majority of the nitrogen in wastewater is discharged to local surface water. The 2006 IPCC Guidelines for National Greenhouse Gas Inventories estimates the emission factor of N_2O to be 0.5 % of the nitrogen content of the effluent (IPCC, 2006). However, the uncertainty is large, with a range from 0.05 % to 25 % given by the IPCC. In the absence of better estimates the IPCC guide was followed and the effect of changes in this factor was tested in uncertainty analysis. According to standards used by Statistics Norway, the ratio between phosphorous and nitrogen is 1.6:12 (SSB, 2010). From the measured phosphorous content, we were able therefore to estimate the amount of nitrogen amount in the wastewater. Treatment efficiencies for mechanical and chemicals plants were taken from Venkatesh and Brattebø (2009).

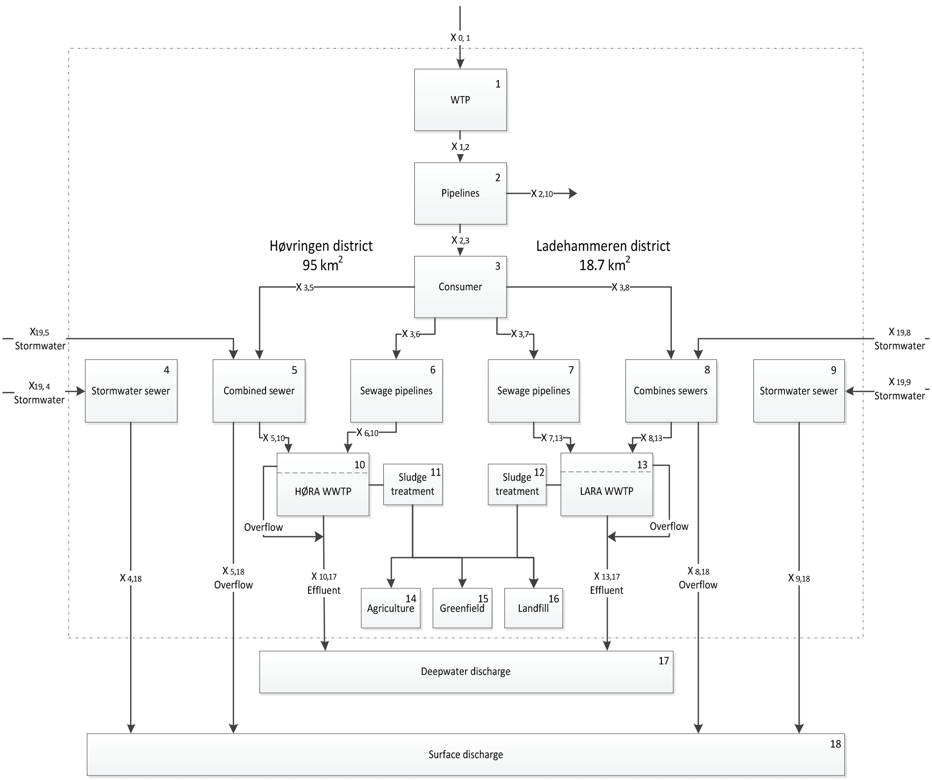


Figure 5. Water and wastewater flow diagram for Trondheim.

There are no nitrification/denitrification processes in the wastewater treatment plants and the nitrogen content of the wastewater is not measured. Nitrous oxide (N_2O) is a potent greenhouse gas with a characterisation factor 298 times that of CO_2 (IPCC, 2007b). In the absence of a nitrification/denitrification process, the majority of the nitrogen in wastewater is discharged to local surface water. The 2006 IPCC Guidelines for National Greenhouse Gas Inventories estimates the emission factor of N_2O to be 0.5 % of the nitrogen content of the effluent (IPCC, 2006). However, the uncertainty is large, with a range from 0.05 % to 25 % given by the IPCC. In the absence of better estimates the IPCC guide was followed and the effect of changes in this factor was tested in uncertainty analysis. According to standards used by Statistics Norway, the ratio between phosphorous and nitrogen is 1.6:12 (SSB, 2010). From the measured phosphorous content, we were able therefore to estimate the amount of nitrogen amount in the wastewater. Treatment efficiencies for mechanical and chemicals plants were taken from Venkatesh and Brattebø (2009).

Scenario building

The assessment of the existing water and wastewater systems in Trondheim showed that environmental impacts per person were low, with the high quality water source, the robustness of the surface waters receiving outflows, and the utilisation of the sludge for biogas production and heat recovery all contributing to this. Nevertheless, carbon neutrality is the desired objective, as well as adaption to a changing climate. Scenarios were therefore built based on the objective of improving the impact compared to the conventional system. Energy use, chemical use and nitrogen emissions are the main contributors to global warming within wastewater systems, and the choice of alternative systems had to be based on improving some of these categories. Alternative solutions, such as rainwater harvesting, greywater recycling, and use of alternative treatment systems such as membrane bioreactors (used at BedZED in England) were excluded, based on findings in the literature.

Scenario A was the business-as-usual water and wastewater system for Brøset. The wastewater was sent to the LARA chemical treatment plant (right side of Figure 5). The installation of water saving appliances and the adoption of water saving behaviours by inhabitants can be effective ways to reduce water consumption in urban areas. According to Butler et al. (2010) it is possible to reduce water consumption to 105 litres per person per day using commercially available appliances (average consumption in Trondheim is 160 litres per person per day). The environmental effect of this measure was tested in Scenario B. All the teams in the parallel commissioning process suggested some degree of local stormwater management, such as use of green roofs, permeable surfaces, systems for increased infiltration and retention. While local stormwater treatment would be challenging due to the cold winter climate and clay soils in Trondheim, it is not impossible. The environmental impact of a system incorporating local stormwater treatment was assessed in Scenario C. Constructed wetlands for greywater treatment are an alternative to the conventional system. Constructed wetlands require only small amounts of energy for operation and the use of chemicals is avoided. This type of system has been successfully installed in the Klosterenga apartment complex in Oslo, Norway (Jenssen et al., 2005). Source separation of wastewater and local treatment of greywater at Brøset was assessed in Scenario D.

In summary, the four scenarios assessed were:

Scenario A: Business-as-usual. Stormwater to the WWTP.
Scenario B: Installation of water saving appliances. Water consumption down from 160 l/p/d to 105 l/p/d.
Scenario C: Local stormwater treatment. No stormwater to the WWTP.
Scenario D: Local grey water treatment in subsurface constructed wetlands. Stormwater to the WWTP.

The lack of data on nitrogen content, the uncertainty in the level of N_2O-emissions, the choice of energy mix and the fact that we had to estimate the share between stormwater and wastewater introduced uncertainty to the results. Uncertainty analysis was performed by changing the electricity mix and the emission factor for N_2O.

For more details about the different systems, see the relevant papers.

Chapter 3

Summary of papers and discussion of main findings

This thesis comprises a collection of papers answering the research questions listed in the first chapter. All the papers involve LCA of infrastructure, applied at either a city or neighbourhood level. Paper 1 assesses alternative household waste systems for the new settlement at Brøset. Paper 2 examines the consequences of an uncertainty factor that has not been adequately addressed in waste management research, namely uncertainty in waste composition. Paper 3 shows how a model combining IOA of household consumption at Brøset with LCA of the waste management system was developed. This was undertaken in order to estimate the full effects of waste prevention. Paper 4 assesses the environmental impact of the water and wastewater system in the municipality of Trondheim. Paper 5 discusses the usefulness of LCA in the early phases of the planning of the new settlement and Paper 6 is a chapter from a book about the planning process for the Brøset development. Paper 6 includes some of the results related to the overall impact from Brøset's infrastructure found in Paper 1 and also additional, comparative assessments of possible alternative water and wastewater systems.

For each paper we will discuss the main findings, the extent of their agreement with the literature and the work of others, and the strength and weaknesses in the results. Recommendations for future work will be discussed in Chapter 5.

Paper 1 - LCA for household waste management when planning a new urban settlement

Paper 1 assesses the waste management system at a neighbourhood level. The objective was to assess the importance of waste management to the overall environmental impact of the development at Brøset, and to compare alternative waste management strategies for the new settlement. The present system in Trondheim is mainly based on incineration with heat recovery and recycling. The business-as-usual waste system was modelled as explained in the methodology chapter. Four alternative scenarios were applied.

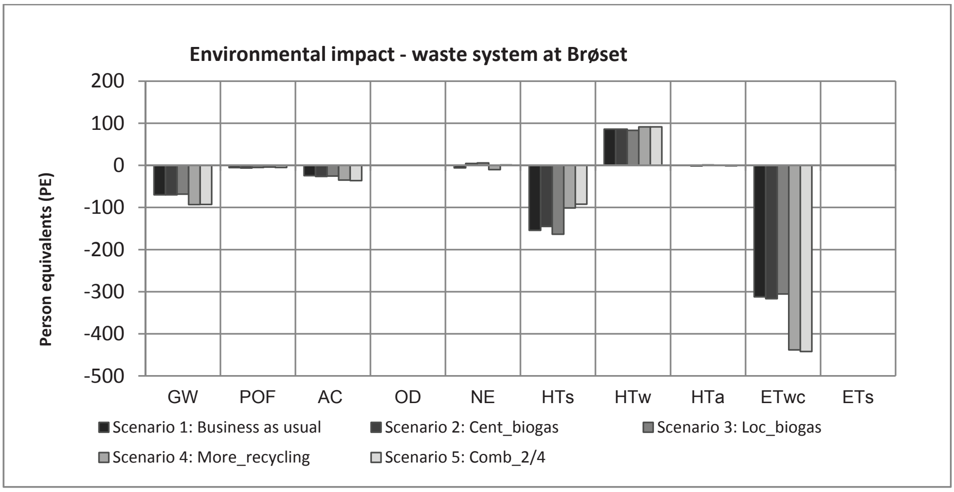


Figure 6. Normalised environmental impact from the five waste management scenarios at Brøset. The impact categories are: global warming (GW), photochemical ozone formation (POF), acidification (AC), stratospheric ozone depletion (OD), nutrient enrichment (NE), human toxicity via soil (HTs), human toxicity via water (HTw), human toxicity via air (HTa), ecotoxicity in water, chronic (ETwc), ecotoxicity in soil (ETs). The system is modeled for 3315 inhabitants.

The results showed that for four impact categories the business-as-usual scenario led to an avoided impact of more than 50 person equivalents (PE) (based on a predicted population of 3315 people). These impact categories were global warming, human toxicity via soil, human toxicity via water and ecotoxicity in water (Figure 6). Human toxicity via water was the only impact category with a net detrimental impact on the environment. This was due to emissions from the incinerator. The largest saving in impact was for ecotoxicity in water, due to substitution of virgin aluminium production with recycled material from both the source-separation of aluminium and the extraction of aluminium from incinerator bottom ash. For global warming the waste management system had a saved impact of 70 PE, or 184 kg CO_2-equivalent per person annually, mainly due to newspaper recycling. The results showed that the waste management system plays a minor role in the total global warming impact from the new settlement. This is, however, not a reason for not optimising the system.

The comparison of the business-as-usual scenario with four alternative scenarios showed the two biogas scenarios to be similar to the business-as-usual scenario, while increasing source-separation ratios led to environmental trade-offs. Increased paper recycling was the main reason for increased saving in global warming impacts in scenarios 4 and 5 (increased recycling and a combination of increased recycling and a centralised biogas system, respectively). For human toxicity via soil the scenarios in which electricity was substituted to the greatest extent were the most environmentally beneficial, while the scenarios substituting the most virgin aluminium were the most preferable scenarios for the ecotoxicity in water impact category.

The choice of technology and energy processes is important for the results of waste management assessments. The challenge with modelling recyclables is that they enter a market-based system, and consequently both the recycling processes and the substituted material or energy can change from time to time. In a study of the global warming impact of the Norwegian waste management system, it was found that metals and plastics should be recycled as materials. There was little difference in impact between incineration and material-recycling of paper and cardboard, and food waste could be treated equally well by either digestion with biogas utilisation or by incineration (Raadal et al., 2009). In a study of various waste treatment options for three Swedish municipalities, it was found that the differences between material recycling, nutrient recycling and incineration were small, but that the recycling of plastics was to some extent environmentally more beneficial than incineration, while biogas production was more detrimental (Eriksson et al., 2005). Tyskeng et al. (2010) reviewed research carried out on several different waste fractions, and found that for paper, plastic, metal and glass recycling was, in general, somewhat better in environmental terms than incineration. A recent study by Merrild et al. (2012) found environmental benefits for the recycling of paper, glass, steel and aluminium when compared with incineration, while the value of recycling cardboard and plastic was more uncertain.

For the glass, metals and food waste fractions, the outcome of this study of Brøset is in line with the literature. The results showed that glass and metals should be recycled and that the impact of food waste sorting is similar to incineration. The gains obtained in terms of global warming impact through increased source-separation of these fractions were shown to be fairly small. For both plastic and paper/cardboard there is some disagreement in the literature. Merrild et al. (2008) found that different combinations of recycling and virgin paper production resulted in relatively large differences in impact, due to variation in the technology and energy sources used. Paper and cardboard constitute a large share of the waste produced in Norway and also have high source-separation rates; it is therefore important to decide whether incineration or recycling is the preferred solution for this fraction. For the incineration process we had fairly good data for emissions, efficiency and replaced energy. For the recycling processes and virgin paper production we relied on the processes available in Easewaste. These processes were chosen based on the destination of the Norwegian paper waste fraction. Norwegian paper and cardboard are mainly recycled within Norway itself; the recycling processes were therefore modelled with the Nordic electricity mix. For this paper we chose to use the default, marginal electricity production mix for the generic virgin paper and cardboard production processes used for substitution. A different option was used in the waste models built for the uncertainty analysis of waste composition and in the hybrid-LCA model assessing waste prevention presented in Papers 2 and 3, respectively. For those models we used the EU-27 electricity mix for the substitution processes. The change from use of marginal energy to average energy was based on the recommendation of the ILCD handbook, which is to use marginal energy only when large changes in systems can be expected (EU JRC, 2010) as a result of the processes modelled. A change in the energy source used for the virgin-paper production process modelled for this paper decreased the savings in global warming impacts, but the system was still beneficial for the environment. This was the effect seen when we changed the technology for the uncertainty analysis, discussed below. Based on the system we have modelled, increased recycling of paper is therefore recommended as a way of reducing global warming impacts. For global warming we knew that the ranking of recycling and incineration with heat recovery was difficult. Findings in the literature and the relatively small differences between the results of the waste management scenarios support this perspective.

In the uncertainty analysis we changed the electricity mix from Nordic to Norwegian and European in all relevant processes. This affected both the total impact from the system and the ranking between scenarios for some impact categories. There was more uncertainty related to the toxicity categories than for the other impact categories. This was due to a lack of characterisation factors for some relevant substances, uncertainties in the characterisation factors that were available, and incompleteness in the normalisation factors for the impact categories (Laurent et al., 2011).

Waste prevention was assessed for this paper using LCA. The result showed less saving in impact because less waste entered the waste system. This was due to the no-burden approach in the model, where impact from the production of goods is excluded. A more advanced model for including waste prevention in the assessment of the waste system was developed for Paper 3. Increasing recycling rates or succeeding with waste prevention will be dependent on the willingness of the inhabitants of the Brøset settlements to recycle or adopt waste preventing behavior. While the Norwegian recycling strategy can be said to be a success, we have not succeeded with waste prevention. Waste prevention will be further discussed in Paper 3 and in the recommendations for further wok in Chapter 5.

The estimated waste composition and sorting efficiencies were assumed to be robust, and uncertainty related to these parameters was not investigated further in Paper 1. A study of the role of waste composition and sorting efficiencies for accounting and comparative LCA is, however, presented in Paper 2.

Paper 2 – Influence of assumptions about household waste composition in waste management LCAs

In Paper 1 we assumed that the data for estimated waste composition, based on waste composition analysis, public data, information from the waste company and data found in the literature, were fairly robust. However, in a comparison of European waste management systems Gentil et al. (2009) found large variations in the literature data on waste compositions. By applying average EU, typical 'northern European' and typical 'southern European' waste compositions, differences in the order of 100–200 kg CO_2 equivalent per tonne waste were found. In a qualitative comparison between different waste LCA models Gentil et al. (2010) found waste composition to potentially have significant impacts on the results of waste system analysis. Dahlén et al. (2009) discussed the many sources of uncertainty in publicly available waste data and in waste composition analysis. In response to the relatively sparse literature in this field, Paper 2 estimates and discusses the theoretical consequences of uncertainty in waste composition. An average waste composition for five cities was estimated and used as a reference scenario for an analysis of the effects of systematically altering the waste composition by ± 15 % for each fraction.

Based on the reasons for performing waste management LCA and the availability of data, there are two methods available for estimating the weight and composition of waste: the use of fixed source separation ratios and the use of fixed waste composition (weight by fraction). These are explained in the methodology chapter, and in Paper 2 itself. The results of the assessments show that global warming, nutrient enrichment and human toxicity via water are the impact categories that are most sensitive to changes in waste composition (Figure 6). The variation in results was larger for the calculation method featuring fixed source-separation *ratios* (Case 1) than for the method using fixed source-separated waste *quantities* (Case 2). The results obtained suggested that changes in the paper and plastic fractions are of most importance for global warming. For nutrient enrichment the food waste content was the most important, while for human toxicity via water the content of aluminium, and most notably the amount of substituted virgin aluminium, was of significant importance. All scenarios involving the increase in the weight of recycled aluminium reduce human toxicity via water impacts, according to the results obtained.

Figure 7 represents the uncertainty involved when performing accounting LCA using the two different calculation methods. However, we also wanted to assess the importance of waste composition for comparative LCA. When we compared different waste management strategies we found the results to be fairly robust, independent of waste composition. For resource depletion and human toxicity via water, increased metal source-separation was clearly shown to be favourable. Increased recycling was shown to reduce impacts in the acidification and global warming categories.

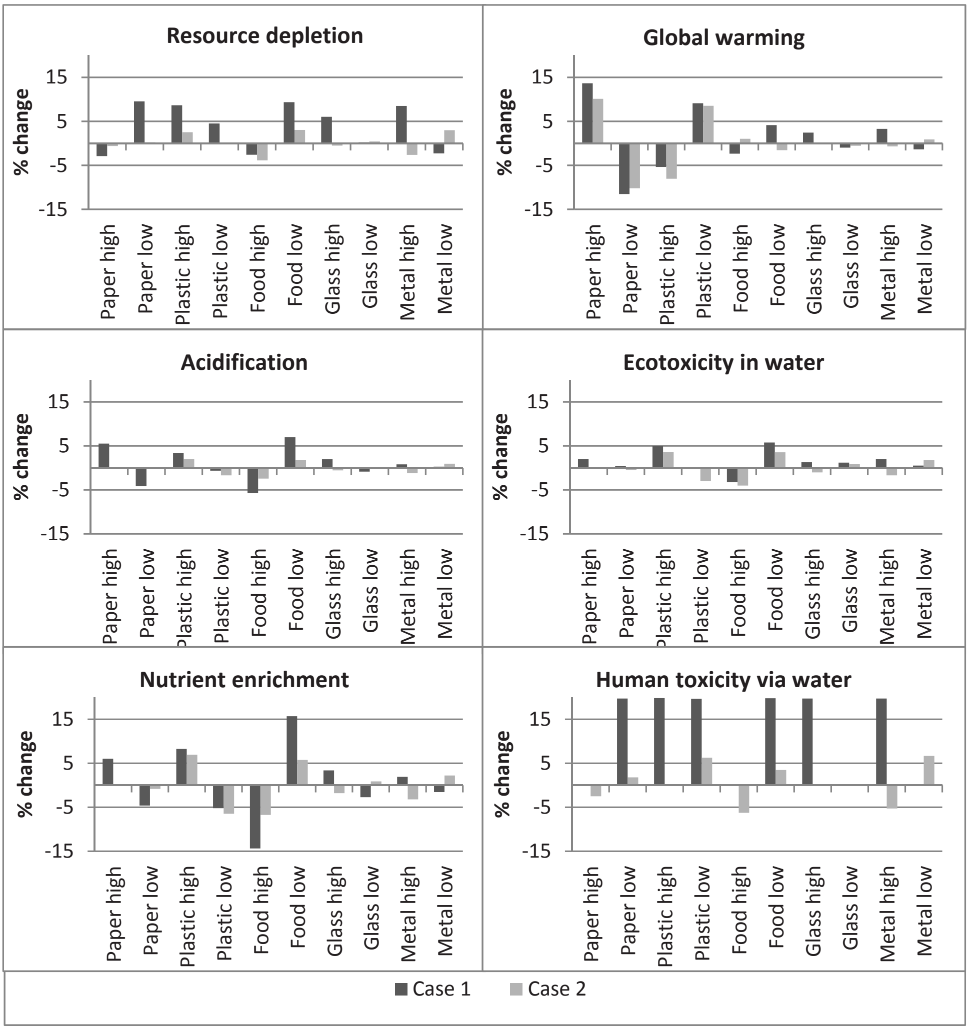


Figure 7. Changes in total impact as result of uncertainty in each waste fraction. A positive bar represents a beneficial change in total impact.

This waste management system modelled for Paper 2 was less complex than the one used to model the Brøset system, presented in Paper 1. Plastic waste, for example, was only used to replace virgin plastic production in Paper 2. In reality plastic can be used to replace wood or as fuel in industry, as included in Paper 1. This makes increased plastic recycling less favourable, at least for global warming; however, the conclusions regarding which measures should be adopted would remain the same.

Large differences in the waste composition between the five cities included in this study were found. Changes in the chemical composition of each waste fraction, or the different share of each fraction in the main fraction, are assumed to be equal between the scenarios. However, the ratio within each fraction of, for example, newspaper and other paper sources, soft and hard plastic, or aluminium and steel, is important for the results of accounting LCA and could also be important for comparative LCA. Larger variation in results could therefore have been expected if uncertainty in these factors had been included in the analysis. We can nevertheless conclude by stating the importance of including system-specific waste composition and, when available, the measured amounts of source-separated waste in future analyses.

Paper 3 – Using IO-LCA to explore how household waste prevention influences economy-wide GHG emissions

In Paper 3 a new model for estimating the effect of waste prevention is presented. The model was based on hybrid-LCA methodology. The use of straightforward LCA, described in Paper 1, to model waste prevention led to less waste entering the system, which in turn resulted in less avoided impact in the global warming impact category. There are two important advantages with the hybrid-LCA model described in the third paper. The first is the inclusion of the upstream effects of waste prevention, based on impacts from consumption, and the second is the possibility to include the rebound effect in the assessment.

The total impact from the reference scenario for waste management system was 4 kg CO_2-equivalent per household, while the total impact related to consumption was 22 746 kg CO_2-equivalent. The main reason for the total impact from the waste management system being environmentally damaging in this assessment, in contrast to the assessments performed in the Paper 1, is that the Norwegian electricity mix was used in this case. The use of this mix reduces the advantages of replacing energy and materials. The Norwegian electricity mix had to be used to model the impacts of the waste system in combination with those from consumption because this was the default electricity mix in the input-output database for Norwegian consumption.

The model was tested on 16 scenarios: a reference scenario, five scenarios including 50 % less food waste, five scenarios including 50 % less textile waste and five scenarios including 50 % less paper waste. The no rebound scenarios were compared with simple rebound, marginal rebound, holiday rebound and restaurant, second hand or cultural rebound (depending on the fractions prevented). Reduction of paper waste is the measure with the largest effect, if only the waste system is accounted for, increasing the total impact from 4 to close to 50 kg CO_2-equivalent per household. When we include the upstream impact, paper waste prevention is the measure with the lowest avoided impact of the scenarios examined (Figure 8).

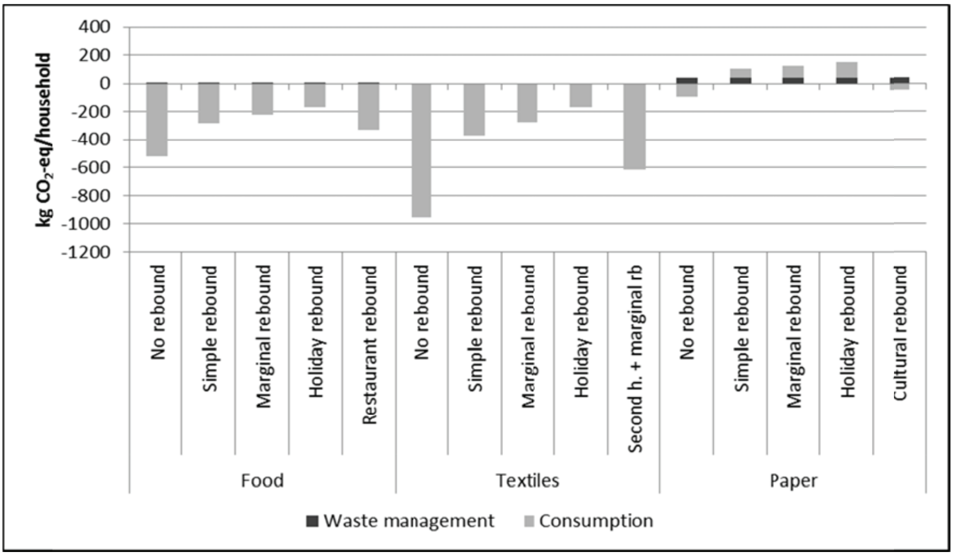


Figure 8. The effect of waste prevention. Change in impact in the total system. All bars are related to the reference scenario. Negative bars indicate improvements in estimated impact compared to the reference scenario.

Prevention of textiles with no rebound effect was the scenario with the largest savings in impact compared to the reference scenario. The rebound effect is, however, important. The results showed that if money saved by the reduced consumption of clothes is spent on a holiday then the beneficial effects of waste prevention will be strongly reduced. If, however, some of the money is used on second-hand clothes and some is spread over other categories (marginal rebound) the saving in impact from this measure would be considerable.

The reduction in food waste resulted in less avoided impact in the no rebound effect scenario than reduction in textile consumption. One important reason for this is the relationship between the amount of food waste and the actual reduced consumption of food. As explained in the paper a 50 % reduction in food waste will only give an 11 % reduction in the consumption of food and drinks products.

In waste management research we often discuss fractions such as paper, plastic, food, metals and glass, and how these fractions should be treated in the best possible way. In the first paper we found recycling of paper to be important for the global warming impact category and metal recycling for some of the toxicity impact categories. In the present paper we found textiles to be an important fraction for waste prevention. It should be noted that the model is not developed to the extent that waste prevention of all fractions can be analysed. We can therefore not at this stage estimate or compare the effect of waste prevention measures for fractions such as plastic, metals and glass. There are also relatively large uncertainties involved in the present study, and none of the values shown in Figure 7 should be taken as absolute values. Nevertheless, this study shows the importance of the upstream activities for the total impact of the system, and how the rebound effect could significantly decrease the benefits of successful waste prevention. When the objective is to reduce the total environmental impact from a new settlement, waste prevention has to be related to avoided impact from upstream activities. Possible savings in impact from the waste management system are of relatively minor interest.

Paper 4 – Life cycle assessment of the water and wastewater system in Trondheim, Norway – A case study

The fourth paper deals with the entire water and wastewater system in Trondheim. As explained in the methodology chapter, most decision support tools made for water and wastewater systems are still focused on the technical aspects of such systems. Although technical optimisation motivated by regulatory issues and economic efficiency is, of course, important, in order to move towards a more sustainable water and wastewater sector the most important environmental impact categories have to identified, together with the processes and parameters causing this environmental impact.

The normalised results of the assessments showed that marine and freshwater eutrophication are the two most important impact categories, with by far the highest impact being marine eutrophication. The reason for the large potential impact on marine eutrophication is the low-grade treatment in Trondheim's WWTPs. These do not have a nitrification/denitrification process, leaving most of the nitrogen in the effluent to be discharged to the nearby seawater fjord. There is work ongoing to improve LCA modelling of regionalised impacts (Finnveden et al., 2009; Gallego et al., 2010). However, impact analysis at this level of detail is not yet available, and local conditions have to be considered separately when interpreting the results. In Trondheim the fjord to which sewage is discharged has been found to be robust (Oceanor, 2003) and marine eutrophication is therefore not a problem. Freshwater eutrophication was found to be the impact category with the next largest impact. The water and wastewater system contributed 3 % of one person's total yearly impact in this category. The impacts here were not directly due to discharges from the sewage system in Trondheim, but stemmed from the use of coal to produce energy in the background system.

Per capita climate change, ozone depletion, photochemical oxidant formation, particulate matter formation, terrestrial acidification, mineral resource depletion and fossil resource depletion impacts due to the waste and wastewater system were all less than 1% of the average impact of one person in Europe (Figure 9). The WWTPs contributed more than 45% of the impact in each category.

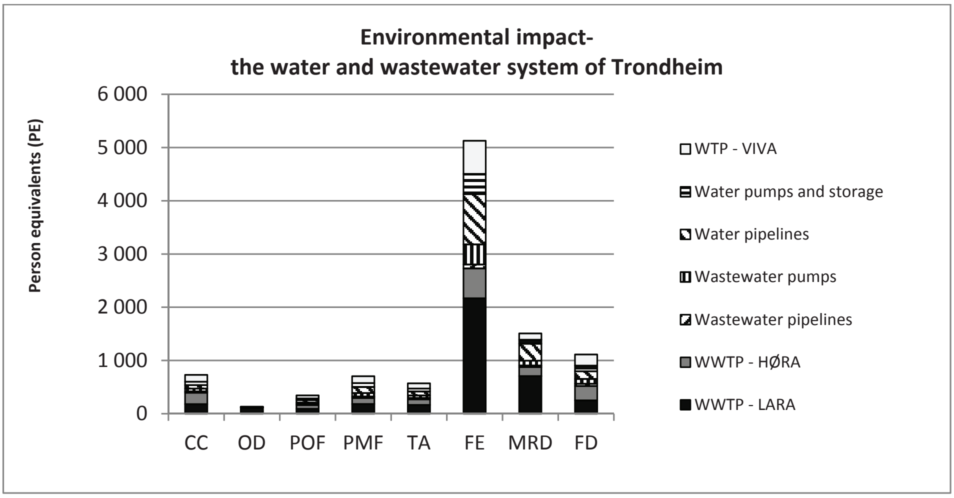


Figure 9. Normalised environmental impact from the water and wastewater system in Trondheim. Related to approximately 170 000 inhabitants. The impact categories are climate change (CC), stratospheric ozone depletion (OD), photochemical oxidant formation (POF), particulate matter formation (PMF), terrestrial acidification (TA), freshwater eutrophication (FE), mineral resource depletion (MRD) and fossil resource depletion (FD). Marine Eutrophication is excluded from the figure.

The climate change impact category is of particular interest in this study. The WWTPs were shown to have the largest impact, with multiple sources, including energy and chemical use (iron chloride), N_2O-emissions and use of materials (Figure 10). The two plants differ due to the different technical solutions employed at each location, and more specifically due to the fact that the plant using chemical decontamination technology (LARA) exports energy to the district heating system. The annual global warming impacts were calculated to be 29 kg CO_2-equivalent per PE hydraulic load or 48 kg CO_2-equivalent per capita. The impacts arising from processes occurring after water is delivered to the consumer were found to be larger than the upstream impacts for this impact category. This is in accordance with findings in Lassuax et al. (2007).

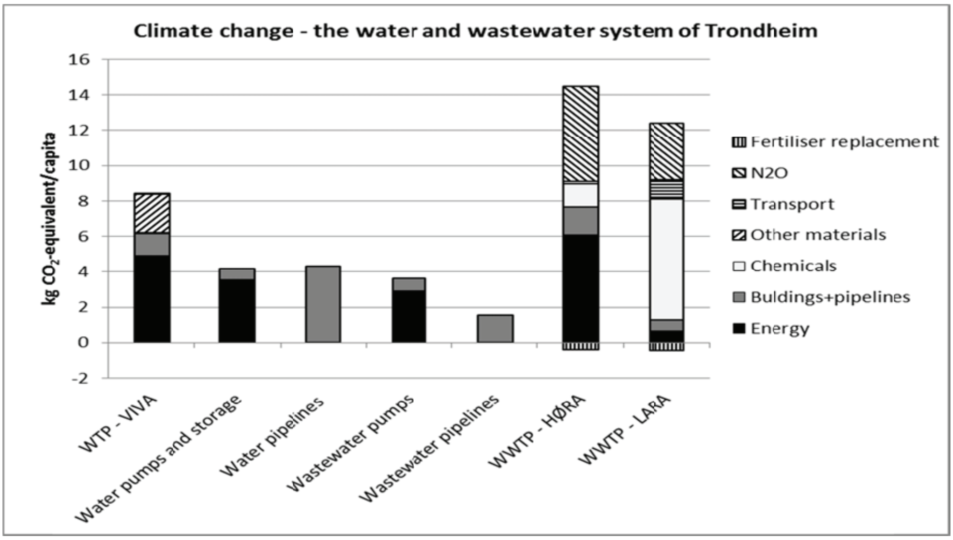


Figure 10. Impact on climate change per person annually from the water and wastewater system in Trondheim.

The United States Environmental Protection Agency assign wastewater as the sixth largest contributor to N_2O-emissions worldwide, accounting for approximately 3 % of N_2O-emissions from all sources (USEPA, 2006). Nitrous oxide emissions formed 18% of the global warming impact from the water and wastewater system in Trondheim. This parameter, however, has large uncertainties, in terms of both the amount of nitrogen emitted and the nitrous oxide produced as a consequence of these emissions. We used the IPCC emission factor for calculating the amount of nitrous oxide, which has an uncertainty range of 0.05% to 0.25 % (IPCC, 2006). The lower and upper limits of this uncertainty range changed the impact per capita to 40 kg and 82 kg CO_2-equivalent annually, respectively.

The sensitivity of the model to changes in the electricity mix was also tested by applying a European mix, which resulted in a doubling of impact per capita. However, neither the change in nitrous oxide emission factor nor the change of electricity mix resulted in the per capita impact of the water and wastewater infrastructure exceeding 1 % of the total global warming impact per person (in Trondheim). For freshwater eutrophication, a shift to a European electricity mix resulted in an impact from the system that was more than 10 times larger than that obtained using the Nordic electricity mix. This was due to the higher share of electricity generated from coal in the European mix. In particular, the disposal in surface landfill of spoilage from the mining of lignite that is used in electricity production was counted as a significant contributor to freshwater eutrophication.

Although the overall impact of the water and wastewater system was found to be fairly small, improvements are possible. However, the use of energy and chemicals is related to treatment efficiencies, and improving one of the parameters could easily result in trade-offs in the system. The major weakness of this study is the lack of data for the nitrogen content of the effluent and the uncertainty in N_2O calculations. The IPCC value is based on weak data according to Kampschreur (2009), and they have questioned the validity of the calculation method. This is for the time being, however, the best available method for including N_2O-emissions in the model.

Paper 5 – Use of LCA to evaluate solutions for water and waste infrastructure in the early planning phase of carbon-neutral urban settlements

Paper 5 discusses the use of LCA in the early stages of the planning phase of the new settlement at Brøset. Brøset was in a very early stage of the planning process when this research project started in 2009, and the municipality is at the time of writing close to finishing the master plan for the area. We had little knowledge about the role of, and the best solutions for, the supporting infrastructure, and the idea was to use LCA to address this problem. LCA has been used in the research community for many years, especially in the waste field. In the course of the last decade water and wastewater systems have also been assessed using this methodology. The problem is not a lack of LCA use in the research field but its implementation in the design process of real projects. The experience from ambitious projects in Norway similar to that at Brøset is that the more environmental objectives put into the planning of new settlement, the more complex the planning phase becomes (Narvestad, 2010). At the same time we knew that the best strategy for environmental protection is to include environmental criteria at an early stage (Farreny et al., 2011b). The importance of prioritising between targets at an early stage therefore becomes important.

The Brøset project began with a desire in the research community to create a carbon neutral settlement (Figure 11). The municipality became interested in the idea, and at the same time a nationwide climate project, ‘Future cities’, was started by the Norwegian Ministry of the Environment. A research project was established with the aim of following and contributing to the planning process of a new settlement. One part of the research project was to answer the research questions found in this thesis. An important part of the planning process was the use of a parallel commissioning process, where four interdisciplinary design teams prequalified for participation in the planning process. Contributions from the design teams were supposed to form the basis for establishing alternative technical solutions for the supporting infrastructure. The level of detail in their submissions was, however, low, and in order to construct the scenarios for this thesis the results from the parallel commissioning process had to be combined with information from the municipality and from the literature. With the help of scenario building and LCA, the total impact from the infrastructure systems, the contribution of the systems to the overall impact of the settlement, the differences in impact between the applied scenarios, and the trade-offs between the impact categories could be discussed. The municipality was informed about the results of the assessments, and was in this way able to incorporate the results in the planning process. The assessments of supporting infrastructure found waste, water and wastewater systems to have little influence on the total environmental impact of the settlement. In addition there were little to be gained from introducing alternative technical treatment systems. However, the assessments showed potential for savings in impact if waste prevention and water saving were to be successfully implemented. Targets for waste prevention and water saving were therefore suggested. The municipality intends to implement local stormwater treatment due to capacity problems in the wastewater network. This was supported by the teams involved in the parallel commissioning process based on the recreational value of water for the local environment. Assessment of alternative water and wastewater systems for the Brøset area also support this measure, with some constraints, as we will discuss in more detail in Paper 6.

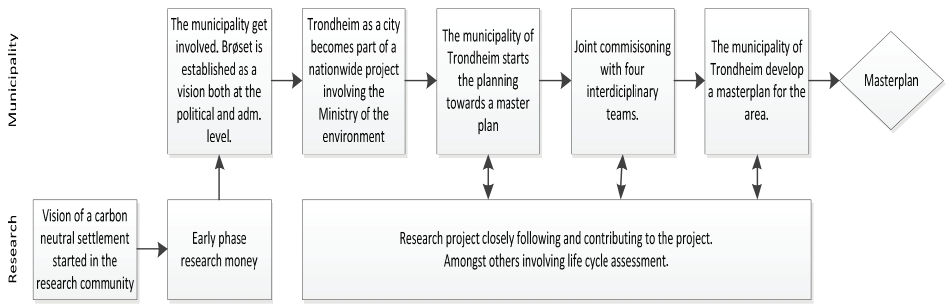


Figure 11. The planning process for the Brøset project, involving both the municipality and researchers.

The opportunities when using the life cycle methodology are many; LCA helps expanding the perspective, often indirect emissions override direct emissions in importance, we get a good overview both of the total impact from a system and where in the system the main contributors to the impact can be found, and we can consider trade-offs between environmental impact categories when comparing alternative systems. This is why LCA can be an important tool even in the early stage planning phase. However, all methodology modelling and assessing real life systems have constraints; the same is true for LCA. It is important to be aware of these constraints. In addition to problems with data uncertainty in this phase of the project results from the assessment pointed at waste prevention and water saving as two measures for reducing the total impact from the settlement, user participation therefore becomes important. LCA, however, is a quantitative method for estimating environmental impact, social and economic implications of the scenarios assessed is therefore not included.

Paper 6 – Environmental impact of water, wastewater and waste infrastructure

Paper 6 is a contribution to a book written about the planning process of the new carbon neutral settlement at Brøset. It will cover the planning phase from the very beginning of the project to the master plan is finished. One of the chapters – this paper – will cover the work package 'Concept for carbon neutral neighbourhoods – Infrastructure'. The results found in Paper 1 will be repeated in this paper, but not discussed here. The alternative water and wastewater systems for Brøset are briefly described in Paper 5, but are more thoroughly discussed in this paper.

The research field of water and wastewater has many levels of detail; we are in this paper concentrating on the research dealing with environmental impact of conventional systems and available alternative solutions. How we decided on the four scenarios included is explained in Chapter 2. The four scenarios are the business-as-usual scenario (Scenario A), the water saving scenario (Scenario B), the local stormwater treatment scenario (Scenario C) and the local grey water treatment scenario (Scenario D). The impact of the business-as-usual scenario is fairly similar to the average impact from the citywide assessment done in Paper 4. The potential impact on marine eutrophication is left out of the Figure, based on the robustness of the recipient as discussed in Paper 4.

The impact on global warming from the system is 11 PE or 36 kg CO_2-equivalent per person annually, compared to 48 kg CO_2-equivalent per person for the citywide system (Figure 12). When we compare the alternative scenarios small improvements in impact can be found for scenario B and C in all impact categories. The largest potential saving is in freshwater eutrophication, where reduced energy and chemical consumption in all the alternative scenarios improve the total impact. Introducing local greywater treatment improves some impact categories while other gets worse. Despite savings in energy and chemical use with this solution, production of light-weight-aggregates, which is used to construct wetlands in cold climates, have an environmental impact in line with the impact saved in the conventional system.

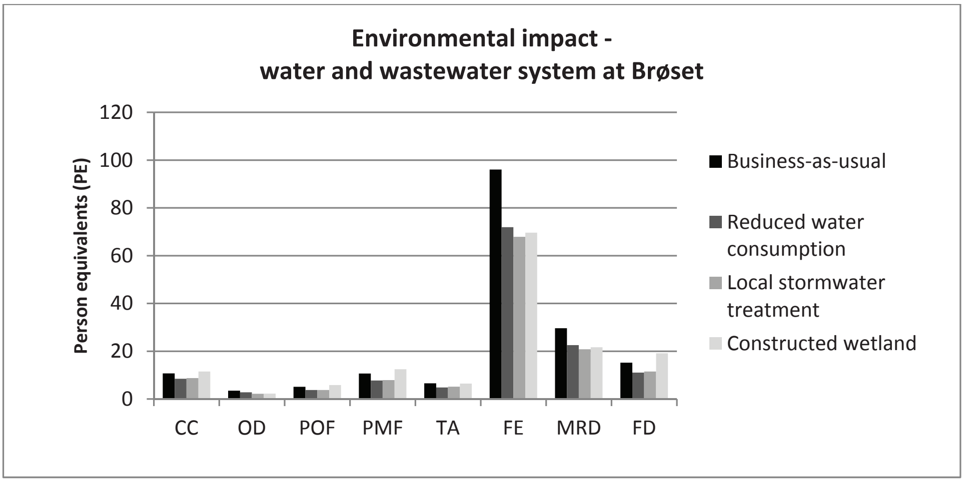


Figure 12. Normalised impact for the four scenarios comparing alternative water and wastewater systems at Brøset. The impact categories are climate change (CC), stratospheric ozone depletion (OD), photochemical oxidant formation (POF), particulate matter formation (PMF), terrestrial acidification (TA), freshwater eutrophication (FE), mineral resource depletion (MRD) and fossil resource depletion (FD). The system is modelled for 3315 inhabitants and some non-residential activity (3500 PE hydraulic load).

If installed on devices using hot water, water saving appliances would, in addition to reducing the impact from the water and wastewater system, reduce the need for energy for water heating. There is also potential for heat recovery from the shower and from other devices, such as washing machines and dishwashers. This was not explored in this thesis. Local stormwater treatment is a solution preferred by Trondheim municipality. There are, however, challenges for this technology due to Trondheim's cold winter climate and clay soils in the Brøset area. It is also important to note that the relatively small potential for environmental savings in the water and wastewater system very easily could be outweighed by impact intensive construction work, if this were needed for the successful implementation of local stormwater treatment.

The limitations of this study lay in the level of detail in the alternative scenarios. The literature review and experience from other settlements left us with the impression that alternative water and wastewater treatment systems were most relevant where water scarcity, or other constraints on water resources or effluent-receiving water bodies, applied (Larsen, 2011). Local solutions for wastewater treatment can be more costly, demand more follow up, reduce the level of comfort for the inhabitants and have higher environmental impacts. Realistic alternative systems were found, but the additional impacts from water-saving appliances, the construction of local stormwater treatment facilities and site-specific details of constructed wetlands were left out of the calculations. This was due to data availability, the level of detail possible at this stage of the planning process and time issues. Water saving measures should be applied and local stormwater treatment could be applied, but it could be more important to optimise the conventional, existing systems in Trondheim through energy and chemical efficiency measures, improving the sludge quality, fixing leakages in the grid system and so on. A study of the Dutch water cycle concluded that it should be possible to create an energy neutral water cycle at a city level (Hofman et al., 2011), while Lundie et al. (2004) assessed the water and wastewater system of Sydney and found energy efficiency, energy generation and additional energy recovery from bio solids to be the most important measures for improving the system.

Contribution of papers

Research question 1
What are the promising concepts of water, wastewater and waste infrastructure design and operation in terms of achieving a carbon neutral settlement at Brøset in Trondheim?

According to the literature, the total impacts on global warming from waste systems are relatively low, and can even be beneficial in many well-developed systems (Gentil et al., 2009). There is therefore little discussion in the literature on alternative waste systems; the discussion is focused rather on the optimisation of the systems found today. One exception is source-separation of food waste and related biogas production. For Brøset both a scenario with source-separation of food waste, a centralised biogas plant and upgrading and use of the biogas for buses, and a scenario with local biogas production with use of the biogas for energy production, were tested in the analysis presented in Paper 1. These alternative systems showed an impact profile very similar to the present system, and connecting to the existing system in town is therefore the best solution. However, phosphorous scarcity might in the future make source-separation of food waste important. Waste prevention (discussed in Paper 3) and increased source-separation of paper and metals were found to decrease the environmental impact from the settlement (or increase the avoided impact) in most impact categories; these are not alternative concepts, however, but constitute rather optimisation of existing systems. Waste prevention was shown to reduce the amount of waste entering the waste system, but also in most cases avoid upstream production. Avoided upstream production could be an important contribution to reduced total impact from the settlement.

For water and wastewater systems, local solutions have been discussed as promising sustainability concepts for future systems (Larsen et al., 2009b; Otterpohl et al., 2003). These include rainwater harvesting, greywater recycling, and local treatment of both blackwater and greywater in separate systems. Rainwater harvesting and grey water recycling have been found, however, to be most important for areas affected by water scarcity, due to energy use, technical requirements and costs (Furumai, 2008; Rygaard et al., 2011; Shirley-Smith and Butler, 2008). The low total impact from Trondheim's current water and wastewater system found in the analysis presented in Paper 4 reduced the list of promising alternative wastewater treatment options to local greywater treatment in constructed wetlands only. In the comparison between a business-as-usual scenario and a scenario with source-separation of wastewater the local wastewater treatment scenario was found to have a higher global warming impact than the business-as-usual scenario, due to the environmental impact of producing the light-weight-aggregates used for constructing the wetland (results shown and discussed in Paper 6). Alternative solutions for stormwater handling and installation of water saving appliances showed more promising results, although it would be important to keep the impact from construction of the stormwater systems to a low level.

Research question 2

How do such concepts contribute to improvements in resource consumption, emissions and life cycle environmental impacts, particularly with regard to greenhouse gas emissions?

Paper 1 showed the importance of today's waste infrastructure system to the overall global warming impact in Trondheim. The Norwegian (and European) strategy of following the waste hierarchy and avoiding the use of landfill has led to the business-as-usual waste system being an infrastructure with beneficial environmental impact. The possibilities for improvement in this system at a neighbourhood level are small. The paper fraction and the replacement of virgin paper production were shown to be particularly important for the saved impact arising from the waste system. We have discussed the importance of the choice of the energy source for virgin paper production earlier in this chapter, and the uncertainty related to this process. The relative low impact (or beneficial impact) found in the business-as-usual scenario was in agreement with literature (Bernstad et al., 2011; Gentil et al., 2009). Waste prevention on the other hand could make an important contribution to reducing total impact on global warming from the new settlement. In Paper 3 we presented a new model for estimating the level of avoided impact of food, textile and paper waste. Food and textile waste was found to be of special interest. However, including the rebound effect in the calculations significantly reduced the benefits of waste prevention.

The existing water and wastewater system was shown to have a detrimental impact. However, this was found to be a very small proportion of the average per capita global warning impact (Paper 4). Wastewater treatment was found to be the process with the largest environmental impact, but contributions from other parts of the system were not insignificant. The contributors to global warming were energy use, the use of chemicals and nitrogen emissions from the effluent. Reduced water consumption and local treatment of stormwater would reduce the impact from the system according to the results presented in Paper 6, but not make it carbon neutral. Lundie et al. (2004) assessed the water and wastewater system of Sydney and found energy efficiency, energy generation and additional energy recovery from bio solids to be the most important measures to improve the system. Health, security, aesthetic value and meeting basic human needs are some of the important aspects of wastewater infrastructure. With Trondheim's current system, the impact on global warming is of less importance in comparison with other impacts from the Brøset settlement. Optimisation should, however, always be strived for.

Research question 3
How can the urban settlements planning process benefit from identification and assessments of such concepts?

The main paper addressing this question is paper 5. The opportunities in using the life cycle methodology are many. LCA helps to expand the perspective of the planning process to include indirect emissions, which often override direct emissions in importance. Total impacts and the location in the system of the main contributors to these impacts can be addressed, and we can consider trade-offs between environmental impact categories by including several categories in addition to global warming in the calculations. Furthermore, LCA and scenario building can be used in combination to compare alternative solutions. For the Brøset case the use of LCA and scenario building helped to reveal the role of the infrastructure, to compare alternative solutions and to decide on possible environmental targets for the area. The results showed that the business-as-usual systems are good solutions, and that no local, alternative solutions for waste or wastewater treatment could improve the system significantly. For the planning process this is important knowledge, and suggests that the main effort should be put into other parts of the project. Local solutions argued for in the literature and tested in projects such as Flintebreite and BedZED should therefore be discarded, and experiences such as that of the BedZED project installing systems that were more energy demanding and more costly than conventional systems thereby avoided (Shirley-Smith and Butler, 2008). While the use of LCA has been important in this planning process, the results showed low total impact from the waste, water and wastewater infrastructure. It might therefore not be necessary to carry out LCA at a neighbourhood level for future projects when well-functioning conventional systems exist. However, if alternative systems are considered for a particular project, LCA is an important tool for environmental assessments. It should also be considered for the analysis of systems at a city, country or conceptual level.

The use of LCA in early-phase planning has advantages, which are discussed above, but also disadvantages. All methodologies used to model real world systems have constraints. This is the case for LCA, and the early phase of planning in this project added extra uncertainty to the assessments. We had to estimate some of the input parameters, such as the waste composition, sorting efficiencies, water use and wastewater production. These input parameters were estimated using average data for Trondheim, analyses of similar density areas in Trondheim, or the literature. What we do not know, and what is very difficult to estimate, is whether these parameters will be different for Brøset compared to conventional neighbourhoods. This area will most probably attract people more willing to adapt to a low-impact society, and we might see a change in habits affecting the input parameters. For the alternative solutions, we had to base our analyses on databases or literature data. These assumptions were therefore not completely adjusted to local conditions. Uncertainty is therefore discussed in relation to each assessment, while Paper 2 is a conceptual study on uncertainty in waste composition.

Chapter 4

Conclusion

This research had several objectives. We wanted to find the role played by Brøset's supporting infrastructure with respect to resource consumption and emissions, particularly carbon emissions, search for promising concepts for infrastructure design and operation that could contribute to achieving a carbon neutral settlement, and investigate how an urban settlement planning process benefits from the identification and assessment of such concepts.

The results of the LCAs of the waste system found the business-as-usual scenario to save impact on global warming. This is in accordance with other assessments of household waste management systems in cities with high recycling rates and energy recovery, such as those by Christensen et al. (2009), Gentil et al. (2009) and Bernstad et al. (2011). The other environmental impact categories assessed were also beneficial or close to zero, except for human toxicity to water, which showed a detrimental impact due to emissions from the incinerator. We found introducing source-separation and digestion of food waste to be similar in impact to the present system. Increasing the source-separation of paper and metal would give increased benefits in several impact categories, with the former being most important for the global warming impact category. The waste system assessed for Brøset excluded source-separated EE-waste, hazardous waste, garden waste and wood waste fractions. Some of these fractions could potentially have a net environmental impact in several of the impact categories.

The water and wastewater system in Trondheim were shown to have a detrimental impact on global warming, but the impact is below 1 % of the annual total impact per person. There were several sources for the impact, such as production of chemicals and electricity and emissions from the effluent. The wastewater treatment plant was found to have the highest impact, but the water treatment plant, the pipelines and the pumps were also shown to make significant contributions to the total impact. All impact categories have a net detrimental impact, with freshwater eutrophication as the potentially most important impact category after normalisation due to the impact from electricity production. The low total impact from the system reduced the number of possible alternative water and wastewater treatment solutions for the Brøset area. Many technologies presented in the literature as having promising sustainability credentials in fact have comparable or higher energy use to the conventional system in Trondheim, and could therefore not be used as alternatives to the conventional system, if carbon neutrality is the objective. Reduced water consumption and local stormwater treatment could, however, be combined with today's system, and their implementation would reduce the total impact from the system, according to our results. Decentralised greywater treatment in constructed wetlands is another alternative wastewater treatment suggested in literature. As a solution that uses less chemicals and direct energy than conventional systems and that has been tested in Norwegian conditions, this alternative provided a suitable scenario to be assessed for the Brøset project. However, the LCA found the production of the lightweight aggregates used for this technology to be energy consuming, making the impact from the alternative system similar to the conventional system. The conclusion of the comparative assessments of water and wastewater systems that there is no alternative system that improves the impact significantly is in line with the conclusion of Remy et al. (2010), who carried out a detailed study of alternative wastewater systems for a hypothetical new settlement.

Assessments of the waste management system and the water and wastewater systems were performed with well-established methods. It was the use of these methods in the early-phase planning of a new ambitious project that was of interest. The combination of system analysis and scenario building proved very helpful in the early stages of the planning phase of the new settlement, for assessing the role of the infrastructure, for including several environmental impact categories, and for comparative assessments of alternative solutions. The results of the assessments can be used to inform important choices, narrow down possible options and highlight where efforts to improve the systems should be made. To narrow down alternative solutions to the conventional systems, literature data and experiences from other ambitious environmental projects were used. Although, system analysis assesses alternative technical solutions, the results of the assessments show the importance of inhabitant behaviour. Increased recycling and waste prevention are based on people being willing and able to increase the source-separation rates and consume fewer goods. Local stormwater treatment is the only measure that would lead to a direct benefit for the people living in the area, if constructed properly. Lifestyle is an important part of the planning for carbon neutrality at Brøset, and this thesis supports the importance of understanding the behaviour of the people who will come to live in this new area.

To perform LCA some methodological choices have to be made and uncertainty has to be accounted for. For the assessments of the waste system we chose to use the Easewaste waste management LCA tool. Knowledge of the impact of Norwegian waste systems and experience of the use of waste-specific LCA tools were limited. Gentil et al. (2010) found that the national origin of LCA models has an impact on the results of LCA on waste systems. Easewaste is specially designed for waste systems and the processes included represent Nordic conditions. These are obvious advantages when applying it to Norwegian waste systems, where the systems are very similar to those found in the other Nordic countries and most of the recycling processes are comparable to those included in the tool. The Easewaste tool was therefore successfully used in all of the assessments of waste management systems found in the thesis, and was found to be very useful in modelling these systems. One disadvantage with the tool is the limited availability of impact assessment methods.

All assessments in the thesis included discussions of uncertainty. However, we found uncertainty in waste composition to be of special interest due to the relatively limited knowledge in the current literature of the effect of uncertainty in waste composition for accounting and comparative LCA. Uncertainty in waste composition was shown to have a relatively large effect on the results in accounting LCA when the sorting efficiencies were held constant. When the amounts of source-separated fractions were known, or when LCA were used to look at the effect of increased recycling or the introduction of alternative technologies, the results were more robust. For the water and wastewater system uncertainty in nitrogen content and the IPCC emission factors for nitrous oxide were examined. Although changes in these parameters were shown to affect the total global warming impact of the system, the total impact will remained low. According to the results from the assessment of the water and wastewater system, marine eutrophication should be a significant problem in Trondheim due to the low efficiency in nitrogen removal. However, the fjord receiving discharged effluence is not sensitive to nitrogen emissions and eutrophication is therefore not a problem in Trondheim. The problem with the use of regionalised impact categories in the absence of regionalised characterisation factors is an acknowledged problem, and work is currently underway in the international LCA community to develop parameterised LCA tools that use regional or local characterisation factors. Until such LCA tools are available, interpretation of LCA results with reference to local conditions is very important.

In order to be able to include waste prevention in the assessments effectively, we found it necessary to develop a new hybrid-LCA model. The model was developed and tested for the first time using Brøset data, and it is the first model, as far as we know, that combines environmentally extended input-output tables, consumer expenditure surveys and waste management LCA to quantify both the upstream and downstream effects of waste prevention. In addition to the inclusion of upstream consequences of waste prevention, one of the most important features of this model is that the rebound effect can be included. Waste prevention is at the top of the waste hierarchy, but has not been successfully implemented in Norway. We found successful waste prevention to decrease the environmental impact from the modelled system in several fractions. The effect of waste prevention was shown to be dependent on the kind of goods avoided, the substitutes used to replace the function of the avoided goods and what the money saved is used for (rebound effect). The rebound effect was shown to have a significant influence on the benefits of waste prevention that could almost offset the effect of reduced production. The model should be developed further to reduce the level uncertainty and to add the possibility of calculating the effect of waste prevention in additional fractions. In addition there is currently too little knowledge of the measures needed to successfully implement waste prevention.

The main conclusion is that the use of system analysis in the early stages of the planning phase of a new carbon neutral settlement can aid the understanding of what the most important contributors to the environmental impact of a system are, and provide decision support for those choosing between alternative systems. LCA is an important tool for assessing, at different levels of optimisation, the alternative solutions suggested in the literature. We found connecting to the conventional systems to be the best option for this new settlement. Greater focus on recycling, waste prevention, water savings and local stormwater treatment were the important factors for reduced global warming impacts from the settlement. The impact from the assessed systems was, however, small compared to the total impact from a residential area.

Chapter 5

Recommendation for future work

The level of detail and accuracy of the studies carried out for this thesis could always have been improved if more time had been available. Some aspects of the systems studied and the results obtained require further investigation and additional research. Waste management systems for the paper, plastic, food and mixed waste fractions are well covered in the literature. There is still not fully agreement between all studies on what is the best treatment option for each fraction, this can often be subscribed to the relatively similar impact of recycling and energy recovery for many fractions. However, the glass and metal fractions are less studied. Glass in Norway is sorted in more than 30 fractions and used to produce of new glass, insulation, building materials and as fill material in road construction. However, the common approach is to assume that the sorted glass is used only for the production of new glass, as in the assessment of the Norwegian waste system by Raadal et al. (2009) and the conceptual analysis of glass recycling by Larsen et al. (2009a). Raadal et al. (2009) considered this assumption to be a weakness of their study. Uncertainty is also found in the metal fraction, both in where the metals are recycled and what they replace. In addition to uncertainty in the technologies used and materials replaced, there is no good data on the share between glass and metal in the source-separated fraction from each city. The same applies to data on the share between different metals. In addition, the extraction efficiency for the metal extraction from the incinerator bottom ash had to be assumed. Based on these data limitations, and the importance of especially metal recycling for some impact categories, we recommend that a specific study on the glass and metal fraction from Norwegian (or other countries') household waste should be carried out.

Waste prevention is a field with a limited research base so far. Waste prevention is at the top at the waste hierarchy. However, the no burden approach to analysis of successful waste prevention in waste management systems in many cities with well-developed waste systems would suggest that waste prevention actually increases environmental impacts. To assess the full effect of waste prevention we have in this thesis described a model that includes the production phase of goods through the application of hybrid-LCA. The relationship between the impact due to production and the impact from waste treatment is, however, complicated and the model has to be developed further in order to reduce uncertainty. In addition it should be possible to model prevention of more fractions. In addition to model development, waste prevention behaviour and effective measures to encourage waste prevention should be studied in more detail. We suggested waste prevention targets for the Brøset area and the area could be used as a case for further research in the waste prevention field.

In this thesis environmental impact of technical systems has been the main focus. However, as we have seen, many of the conclusions are effected by and depend on the habits of the inhabitants. To succeed with increased recycling people need to be willing and able to separate more of their waste. Although some water savings can be achieved by installing water saving fixtures, in order to reduce consumption to 105 l/p/d, water saving behaviour has to be adopted by the inhabitants. More research should be performed on effective measures for increased source-separation and water saving, hot water in particular.

Environmental impact is only one of the aspects of sustainable development. Social and economic assessment therefore could be added to the analysis of the Brøsets infrastructure. Local stormwater treatment in a neighbourhood of Brøset's size is relatively new in Norway, and in addition to the challenges related to the cold climate and clay soils, there are questions related to environmental, social and economic costs that are not well covered in literature. Although multi-criteria analysis has been applied on a stormwater disconnection project in the south of Norway (Lindholm and Nordeide, 2000) more knowledge is needed in order to draw conclusions on the sustainability of local stormwater treatment systems in urban areas with a cold climate. Due to the lack of experience with such systems in Norway, however, performing multi-criteria analysis that is sufficiently accurate to support decision-making in this early stage of the planning phase of the Brøset development would be challenging.

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Paper 1

LCA for household waste management when planning
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LCA for household waste management when planning a new urban settlement

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ABSTRACT

When planning for a new urban settlement, industrial ecology tools like scenario building and life cycle assessment can be used to assess the environmental quality of different infrastructure solutions. In Trondheim, a new greenfield settlement with carbon–neutral ambitions is being planned and five different scenarios for the waste management system of the new settlement have been compared. The results show small differences among the scenarios, however, some benefits from increased source separation of paper and metal could be found. The settlement should connect to the existing waste management system of the city, and not resort to decentralised waste treatment or recovery methods. However, as this is an urban development project with ambitious goals for lifestyle changes, effort should be put into research and initiatives for proactive waste prevention and reuse issues.

1. Introduction

In the city of Trondheim, located in central Norway, a new urban greenfield settlement with the aim of becoming climate-neutral, is in its planning stage. To achieve climate-neutrality, every process in the construction and operation of the settlement has to decrease its greenhouse gas (GHG)-emissions. Existing infrastructures are usually preferred when planning for a new settlement. But at times, the need for sustainable solutions calls for changes in these systems, and numerous ideas for alternative solutions are brought to the table.

The Norwegian waste policy has been guided by the waste hierarchy – reduction, re-use, recycling, incineration, landfilling – since the early 1990s, and landfilling of organic wastes was banned in Norway in 2009. The waste hierarchy has also been focused upon in waste management research, and has been validated as a rule of thumb by Finnveden et al. (2005). The ranking of recycling and incineration with heat recovery, and deciding where to include biological treatment, has, however, been difficult (Finnveden et al., 2005). Waste management systems are found to be responsible for approximately 2% of the total GHG emissions – originating primarily from organic waste in landfills and the incineration of wastes with a fossil origin (McKinsey and Company, 2009). In a study of 40 generic systems in Europe, the sorting efficiency of paper, energy substitution and binding of biogenic carbon in landfills were found to be the most important factors for GHG-emissions, with all the systems representing savings to the environment on this impact category (Christensen et al., 2009). According to Astrup (2011), wastes should be perceived as resources. Though he is particularly concerned about the carbon content of the wastes, the advocacy of this perception is supported by the energy value and the possibility of harnessing the material/nutrient value of the wastes, as well.

Waste prevention is defined as the reduction and reuse of waste and covers the two measures at the top of the waste hierarchy. A Norwegian Official report on this issue was published in 2002 (Ministry of the Environment, 2002), but while Norway already is close to its goal of 75% recycling or energy recovery, the amount of household waste has been steadily increasing, and waste prevention is not successfully implemented and experienced in practice. At the time of writing, household waste production stands at approximately 420 kg per person per year. Wilson et al. (2010) calls for attention to the waste prevention issue, due to the fact that in many countries (like in Norway), high recycling rates and the use of incinerators with energy recovery are already well-entrenched. Little research has, however, been carried out in this respect, and it is difficult to find examples on waste prevention achieved both in new and established settlements. According to Sharp et al. (2010a) the potential for waste prevention is assumed to be around 0.5–1 kg/household/week, with the greatest potential realised with a focus on the fractions food waste, garden waste and bulky waste. There are some food waste prevention campaigns that have reached 1.46 kg/household/week (‘Becoming a committed food waste reducer’) and 2.5 kg/household/week (‘Love food Champion’). The results from these kinds of campaigns, and their generalizability, are discussed, however, because of small sampling sizes and specially recruited people (Sharp et al., 2010b).

This study is based on the imperativeness of taking the right decisions when planning for an ambitious new project in an existing town. Life cycle assessment (LCA) is seen as an important

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contributor to getting a systematic environmental assessment of the waste management system (Ekvall et al., 2007). It has been performed in the field of waste management for countries (Bjarnadóttir et al., 2002; Raadal et al., 2009), cities (Cherubini et al., 2009; Larsen et al., 2010) and waste fractions (Astrup et al., 2009; Damgaard et al., 2009; Larsen et al., 2009; Merrild et al., 2009). We have, however, used LCA in an early stage planning phase of a new settlement in the built environment, to analyse different waste treatment scenarios comparing business-as-usual to other centralised and local solutions. We have also discussed the potential for waste prevention, following the waste hierarchy.

2. Methodology

Brøset is a 35-hectare (350,000 square metres) suburban site, 4 km from the city centre in Trondheim, Norway. We can call it a 'greenfield' development project, since the site has not previously been developed for urban usage. There are plans for 1200–2250 new dwellings built in an urbanised and sustainable way, likely to house approximately 2500–5000 people. There are several LCA tools available, but we decided to use the EASEWASTE software, which is specially designed for waste management systems. It is developed by the Technical University of Denmark (Kirkeby et al., 2006), and has been commonly used in similar studies in Scandinavia recently. The scope of the LCA case study is to assess different scenarios for waste management at Brøset, and the functional unit is "collection, transport and treatment during one year, of the waste streams of mixed waste, paper, plastic, glass and metals from 1500 new households (3315 persons) at Brøset in Trondheim, Norway". Source-separated waste fractions as EE-waste, hazardous waste, textiles and garden waste are not considered. The same scenarios will also be assessed for waste prevention. We are taking a no-burden approach excluding all embodied energy of the waste from the calculations. This is a common approach to take when looking at the waste management system in itself (Bjarnadóttir et al., 2002).

All the waste streams are followed to their end destinations through system expansion; thereby accounting also for the avoided emissions and resource use from substituted energy and/or materials. Norway, with a substantial amount of hydropower, has a fairly-clean electricity production, but as Norway is part of the Nordic electricity market, the authors have opted for the Nordic Electricity mix in this study. The effect of this choice is tested in the sensitivity analysis.

The impact assessment method used is EDIP 1997 (Wenzel et al., 1997) with the impact categories Global warming, Photochemical ozone depletion, Acidification, Stratospheric ozone depletion, Nutrient enrichment, Ecotoxicity via water and soil, and Human toxicity via water, soil and air. In addition, Resource Use is included for fossil fuels, metals and some other resources, such as phosphorous. Environmental impacts are normalised according to EDIP97 values of global or EU-15 annual environmental impacts of one person, and the results are given in person-equivalents (PE). The normalisation factor for Global warming is 8700 kg CO_2-eq per person annually. Normalisation factors for other impact categories can be found in Christensen et al. (2007). Resource use is normalised against the average European resource use of one person in 2004. Impact from collection and transportation is included through the entire system.

2.1. Waste management scenarios – incineration, recycling and digestion

The waste management system in Trondheim is based on incineration with heat recovery, with the incinerator being the main heating source for the district heating system in town. The incineration plant treats approximately 200,000 tonnes of wastes annually from Trondheim and the surrounding region. The energy efficiency is 86%, and it delivers 380 GWh heat for approximately 6000 residential buildings and 600 public buildings in Trondheim, covering 30% of the heat demand in the city (TEF, 2010). The district heating system in Trondheim substitutes other heat sources with an energy share of 72.4% electricity, 18.5% fuel oil, 5.2% wood and 3.9% natural gas. Paper, cardboard and plastic are source-separated with kerbside collection. The paper and cardboard fractions are taken to a Material Recovery Facility (MRF) in Trondheim where it is sorted into four fractions, while the plastic fraction is collected, compacted and reloaded before it is sent by trains to an MRF in Germany. Glass and metal are brought to collection points. After collection these fractions are transported 600 km to a glass and metal MRF in Southern-Norway, where aluminium and steel are sorted out and delivered to a recycling facility, while glass is crushed and sorted into many fractions based on colour and quality.

There are three important premises for the choice of scenarios which will impact the result; these are technological solutions, sorting efficiency and the choice of substituted processes. The original waste composition is held constant in the scenarios. The waste composition and sorting efficiency are estimated based on waste analysis of areas of comparable density in Trondheim (NORSAS, 2007), average data from the whole of Trondheim and communication with the waste-handling company in question. Out of the estimated 929 tonnes of waste generated annually by the 1500 households in the new settlement, 25% are assumed to be food waste, 34.7% recyclable paper and cardboard, 10.1% recyclable plastic, 5.7% recyclable glass and 1.4% recyclable metals. The technological solutions are given for the incinerator and for the MRFs which sort the waste, but for anaerobic digestion and treatment of the recyclables, the technology is based on literature data and processes available in EASEWASTE. The processes in EASEWASTE represent Northern-European technology, the same holds for the compensatory processes. Composting is, according to literature, not a good solution for food waste treatment compared to incineration and anaerobic digestion, due to the lack of energy recovery, at least in Northern-Europe where heat can be utilised in buildings (Eriksson et al., 2005). Composting is therefore not considered in this study.

Anaerobic digestion is not new anymore, but it was not until the mid-90s that this was considered to be a tried-and-tested, fully-proven technology for waste handling (De Baere, 2000). In Norway, there were about 10 biogas plants for treating municipal solid wastes in 2009, biogas plants are more common in other North European countries like Germany, Sweden and Denmark. The municipality of Trondheim is, at the time of writing, investigating the consequences of building a co-digestion plant treating approximately 20,000 tonnes of organic waste annually. The biogas will be upgraded and used as fuel for buses in the city. Small-scale biogas plants for treating organic household wastes are not very common in the developed world, and the literature usually deals with biogas plants treating more than 10,000 tonnes/year for household waste (Borjesson and Berglund, 2006; Pöschl et al., 2010). Digestion of food waste in combination with blackwater and small-scale combined heat and power plants (CHPs) has, however, been tested in ecological settlements like Flintenbreite in Germany (GTZ Ecosan project, 2005). This could be a solution for Brøset as well.

While introducing source separation of food waste will change the technological assumptions for the system, another solution could be to increase the share of source separated wastes. Based on the existing waste management system in Trondheim, current plans for future development and available literature, we have developed five scenarios (combinations of technical solutions) to be examined by LCA.

- Scenario 1: Business as usual (connecting to Trondheim's existing waste management system).
- Scenario 2: Centralised biogas plant (introducing source separation of food waste).
- Scenario 3: Local biogas plant (introducing source separation of food waste).
- Scenario 4: Increased recycling.
- Scenario 5: Centralised biogas plant and increased recycling.

Hence, there is one business-as-usual solution and four alternative solutions. Each scenario is explained in greater detail in the paragraphs that follow, and in Fig. 1. Table 1 lists the source-separation efficiencies for each scenario, in addition more information can be found in the Supplementary data.

2.1.1. Scenario 1 – business-as-usual

This scenario is built up as today's system in Trondheim (Scenario 1 in Fig. 1). Mixed waste, 69% of the waste from the households, is collected and sent to the waste incinerator 12 km from the Brøset area. Disposal of residues from the incinerator are accounted for together with aluminium extracted from the bottom ash. Source-separated paper and cardboard are sorted in four fractions, two of them recycled in Norway, one fraction sent to Europe/Asia, and a small residual fraction is incinerated. Source-separated plastic waste is assumed to be sorted into 5 fractions based on available data on the different plastic fractions and an article by Astrup et al. (2009) assessing plastics recycling. For the source separated glass we assume substitution of virgin glass production, this assumption has been made in other studies as well (Larsen et al., 2009).

2.1.2. Scenario 2 and 3 – sorting of food waste with anaerobic digestion

If food waste sorting and a centralised biogas plant are realised in Trondheim, a colour bag system will be introduced due to the limited space available for waste collection containers. Introduction of food sorting, a colour bag system and a centralised biogas plant with substitution of diesel is investigated in Scenario 2. In scenario 3, decentralised treatment of source-separated food is introduced. Biogas production will be used in a CHP plant in the settlement. As this is a decentralized system, introducing one more fraction is not seen as a problem to the space availability, and a colour bag system is not necessary. The inventory data on the anaerobic digester is a combination of an EASEWASTE process on anaerobic digestion and data from Berglund and Börjesson (2006) on energy use in small- and large-scale digesters. The digestives are assumed to replace mineral fertilizer in both scenarios and the sorting efficiency is set to 70%, with 10% of this sorted out in the pre-treatment phase and sent for incineration.

2.1.3. Scenario 4 – increased recycling

When planning for a sustainable new settlement, there is potential for increasing the sorting efficiencies this scenario therefore has the same processes as scenario 1 but with better sorting efficiencies for the fractions paper, plastic, glass and metals. The paper, glass and metals sorting efficiencies are in this scenario, set to 90%. For the plastics fraction, a sorting efficiency of 70% is assumed. We assume the source-separated plastic to have higher quality than in scenario 1, 2 and 3. These are all ambitious goals; even though 55% of the wastes are still left for thermal treatment.

2.1.4. Scenario 5 – centralised biogas plant and increased recycling

In this scenario, we are combining scenarios 2 and 4 with both food waste sorting with treatment in a centralised biogas plant and increased recycling of the fractions paper, plastic, glass and metals.

2.2. Waste prevention – reduction and reuse

Brøset as a sustainable new settlement will focus on lifestyle change, and waste prevention will be an important part of the lifestyle discussion as it is linked to consumption patterns. We want to look at the effect of waste prevention on the waste management system, and will therefore assume the waste prevention potential of the area and use this in the same scenarios as above. As referred to earlier, food is a promising target for waste prevention. About 25% of the food produced is thrown away, and food waste accounts for one-fourth of the wastes entering the Norwegian waste management system (ForMat, 2011). In Norway, there is an ambitious project of reducing the usable food thrown away by 25% by 2015. This takes into account the food wastes generated all the way from production to final consumption in households. Brøset is a similar ambitious project, the authors therefore assume that it is possible to reduce generation of food wastes from the households by the same magnitude, a 25% reduction of food waste entering the waste management system from the households was therefore assumed. Another promising fraction for waste prevention is paper waste, which comprises 35% of the waste entering the waste management system at Brøset. We know that a large portion of this is newspapers and advertisement-material (pamphlets etc.), and it should be possible to reduce these amounts a lot by campaigns and no-advertisement stickers. We assume that we are able to reduce this fraction by the same share as food waste, namely 25%. For the other fractions, a general waste prevention of 5% is assumed. Courtesy this reduction in wastes, one could reduce the specific waste generation from 289 kg/household/year to 240 kg/household/year, and also change the waste composition in the process. To avoid changing the functional unit, we have to consider the prevented waste as a virtual waste flow with no environmental burden and with no transformation in the waste management system as explained in Gentil et al. (2011). How we manage to reduce the waste entering the waste management system is not considered, nor is the rebound effect that is likely to occur as households may spend savings on other products, the management of reuse activities (second hand stores etc.) or possible changes in hazardousness of the waste.

3. Results

Based on the input values, the five waste management systems (scenarios) are modelled, with results normalised and given as person-equivalents (PE) both for environmental impact and resource use. Positive values describe a load to the environment or resource use, while negative values show savings. Four impact categories have an impact or saved impact of more than 50 PE (an impact of 50 PE is 1.5% of annual impact per person in the given category), this is Global warming, Human toxicity via soil, Human toxicity via water and Ecotoxicity in water (Fig. 2). Other impact categories investigated have close to zero impact. Human toxicity via water is the only impact category with a load to the environment, due to emissions from the incinerator. The largest savings in impact is for Ecotoxicity in water and this is due to substitution of virgin aluminium production. For Global warming the waste management system has a saved impact of 70 PE, or 184 kg CO_2-eq per person annually.

When we compare the five scenarios only three impact categories have a change (compared to business-as-usual) of more than 20 PE, this is Global warming, Ecotoxicity in water and Human toxicity in soil. A change of 20 PE is the same as a saved or increased impact per person of 0.6%. For Global Warming, scenarios 4 and 5 with increased recycling are better than the other three, which are almost equal. The main contributor to this difference is the effect of

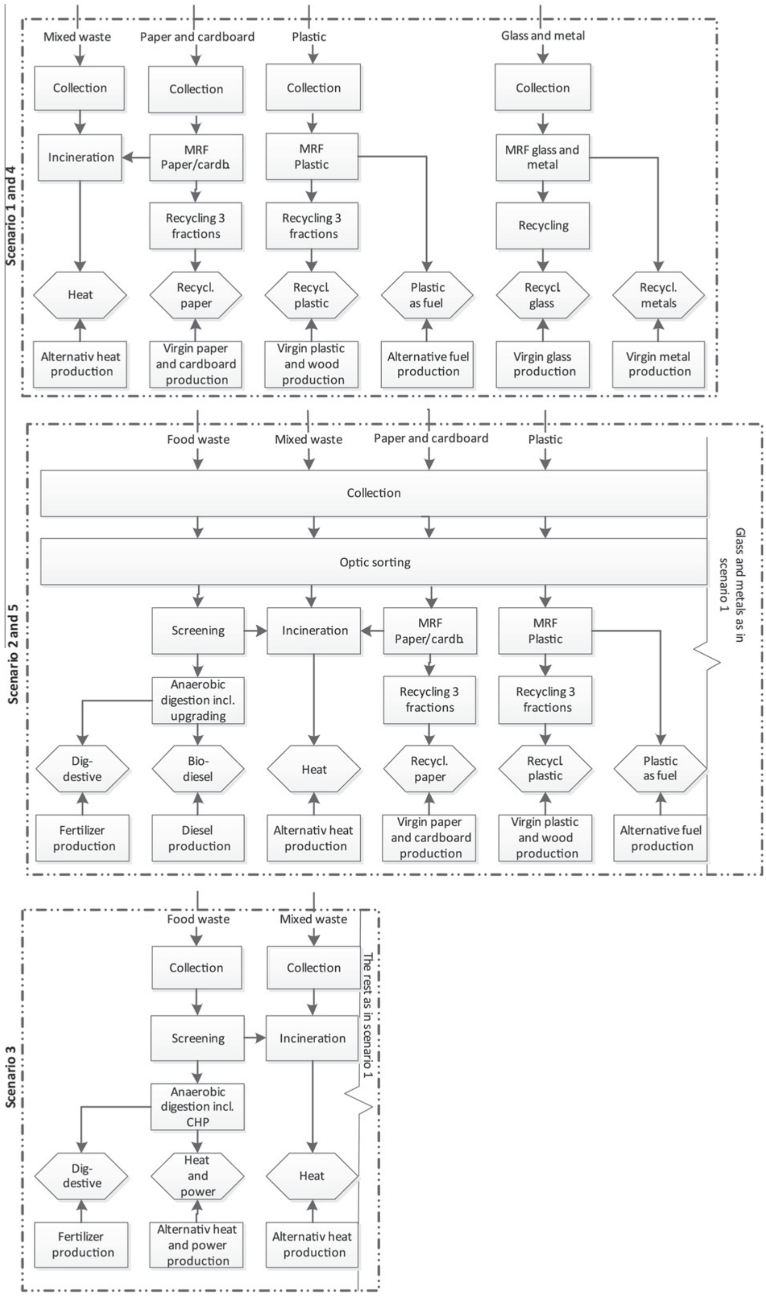

Fig. 1. Five scenarios for waste management of the household waste of a new settlement. Transportation is included in each step.

increased recycling of paper, because the virgin paper production is more energy-intensive than the recycling process. Even if some improvement can be seen by increasing the sorting efficiency, changes are small – only 23 PE or 60 kg CO_2-eq per person a year. For Human toxicity via soil, the largest environmental savings are for the business-as-usual scenario and the local biogas plant. This is due to the importance of replacement of electricity in this impact category, favouring the two scenarios which substitute the most electricity. For Ecotoxicity in water on the other hand, scenarios 4 and 5 are the best options, as in the case of Global warming. Glass and metals recycling contribute the most to these savings in impacts, with recycling of aluminium, reducing PAH (Polycyclic Aromatic Hydrocarbons) emissions from virgin production as the most important parameter. For scenario 4, increased savings are

Table 1
Sorting efficiencies (%).

	Food	Paper	Plastic	Glass	Metal	Others
Scenario 1						
Incineration	100	27	78	41	73	100
Recycling	0	73	22	59	27	0
Scenario 2/3						
Incineration	30	27	78	41	73	100
Recycling	0	73	22	59	27	0
Digestion	70	0	0	0	0	0
Scenario 4						
Incineration	100	10	30	10	10	100
Recycling	0	90	70	90	90	0
Scenario 5						
Incineration	30	10	30	10	10	100
Recycling	0	90	70	90	90	0
Digestion	70	0	0	0	0	0

calculated to 129 PE a year compared with scenario 1. The glass and metal recycling fractions are, however, the fractions with the highest uncertainty based on data availability. These fractions also exhibit large differences in source separation efficiency between the business-as-usual scenario and the assumption of 90% source separation in scenario 4. This does not violate the fact that aluminium recycling is important, and as shown by Damgaard et al. (2009), recycling of aluminium is always favourable even if the amounts are small, because of huge difference in energy intensity of virgin aluminium production vis-à-vis the aluminium recycling process. Collection and transportation had little influence on the results; there are relatively short distances between most of the processes.

Use of non-renewable resources is important from an environmental perspective, and resource use in each of the five scenarios is shown in Fig. 3. Resources with an inter-scenario PE-value change not exceeding 20 are not indicated. We can see from the resources included in the figure that the changes between the scenarios are larger than with the environmental impact categories represented earlier. Scenarios 4 and 5 are the best options for all resources except nickel and phosphorus. Nickel is used in electricity production, and the more electricity replaced the better, making scenarios 1 and 3 the best options for this resource. For phosphorus, all scenarios with anaerobic digestion and use of the digestive as a fertilizer, give savings in virgin phosphorus use. These savings are, however, small – just 39 PE. For the categories hard coal, natural gas and crude oil, the increased recycling of paper and plastic, and thereby replacement of virgin production, are the most important contributors. For aluminium, the authors have assumed a better recycling rate when the aluminium is source-separated than recycled from bottom ash and therefore more virgin aluminium is substituted in scenarios 4 and 5.

Waste prevention is in our study calculated in two ways, one with attention on some waste fractions, changing the composition of the waste entering the waste management system, and one where the waste reduction is evenly distributed between the fractions. The waste amounts entering the waste management system are the same for these two waste prevention scenarios. If we introduce waste prevention into the system, with 25% less food and paper waste, and a 5% general reduction in waste amounts, savings in the waste management system is reduced for the impact categories Global warming, Human toxicity and Ecotoxicity in water (Fig. 4). These reduced savings are due to less waste entering the system. If we compare this scenario with a general reduction at the same range, but without the composition change, the savings will be further reduced for the impact categories Human toxicity and Ecotoxicity in water. For Global warming a general waste reduction is better than a 25% reduction in paper waste, because paper recycling is beneficial for this impact category. Waste prevention can, based on these results, not be argued from benefits in the waste management system.

4. Sensitivity analysis

When we perform a sensitivity analysis we find incineration, together with paper and aluminium recycling to be the most important contributors to the outcome. To test how important these processes are, both for the total impact and for the ranking of scenarios, some key parameters included in the different processes are systematically changed. The choice of substituted energy is affecting the result in waste management systems where district heating systems are replacing other energy sources. In this study, electricity is the main energy source substituted with a share of 72.4% and the choice of a Nordic electricity mix is debatable. We therefore run the model with a 'cleaner' electricity mix (Norwegian, mainly based on hydropower), and a more 'dirty' (European) to see how this change influences the end-results. As we can see from Fig. 5,

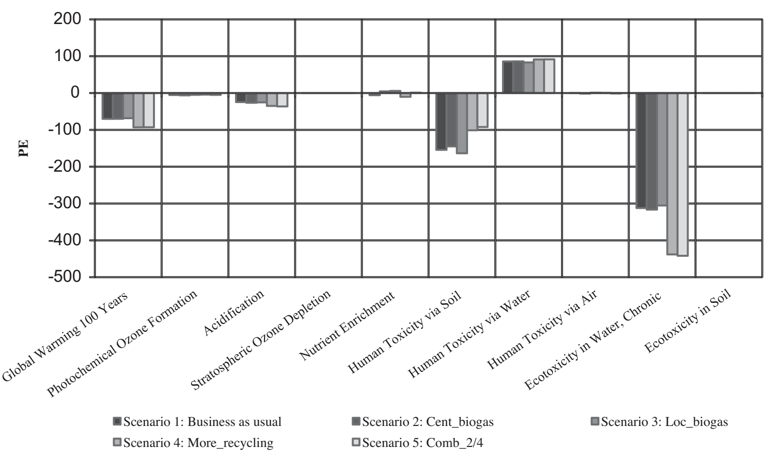

Fig. 2. Environmental impact. Changes in environmental impact compared to the business as usual scenario.

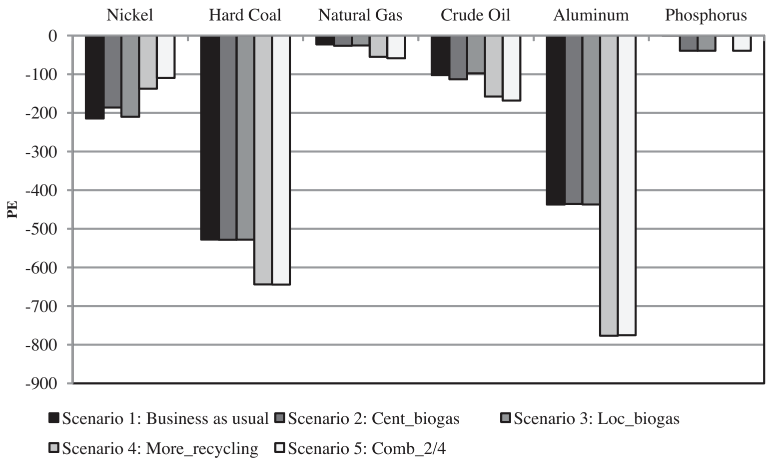

Fig. 3. Resource use. Resource use in the five scenarios, resources with less difference between the scenarios than 20 PE is not shown.

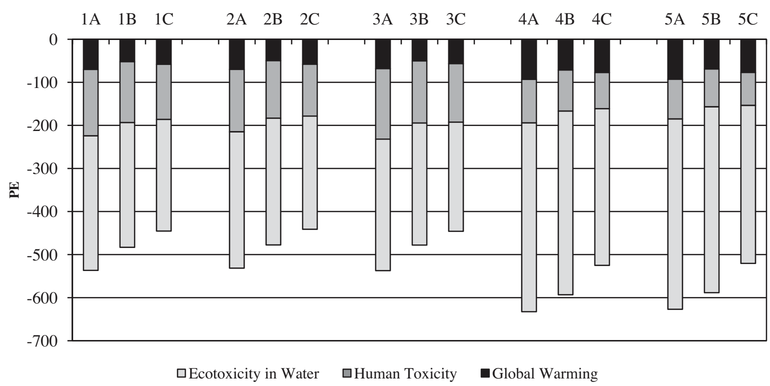

Fig. 4. Waste prevention. Comparing original scenarios (A scenarios) with the same scenarios including waste prevention and composition change (B scenarios) and waste prevention without composition change (C scenarios).

this change has an impact on the savings across all scenarios and categories. Substituting for the Norwegian electricity mix gives less savings, vis-à-vis doing so for the European electricity mix. The ranking between the scenarios is important, while the ranking is fairly robust for Global warming, Human toxicity via soil and Ecotoxicity in water are sensitive to changes in the electricity mix. For Human toxicity via soil the use of a Norwegian electricity mix will give close to zero savings in impact for all the scenarios, while for Ecotoxicity in water it is the European electricity mix that alters the ranking of the result. A more emission-intensive electricity mix favours incineration and energy recovery because of the substitution of electricity by the district heating system.

As shown in Merrild et al. (2008), technology choices influence the results as far as the paper recycling process is concerned. A change from Nordic technology to a more generic one has been tested for paper recycling and virgin paper production. For Global warming, this change led to less savings in the system, but the ranking of the scenarios were the same. For Human toxicity in soil and Ecotoxicity in water, changes in paper technology did not affect the results. The difference between the scenarios is the same for all the impact categories, and the choice of paper technology is therefore fairly robust in this case.

For aluminium recycling, a change in recycling efficiency is tested, with a less-efficient recycling process for the source-separated aluminium included in the calculations. This change only affects Ecotoxicity in water, but the fact that the difference between the scenarios becomes very small is noteworthy. Aluminium recycling is uncertain because of many assumptions included in the assessment of amounts and source separation efficiency. A change in technology also has an effect on the ranking of the scenarios. More thorough studies ought to be carried out on this fraction to improve the accuracy of input data. The Ecotoxicity in water category seems to be the least-robust category giving changes in the ranking of scenarios both for different electricity mixes, as well as the aluminium production/recycling technology.

5. Discussion

In a study of the Norwegian waste management system concerning global warming issues, it was found that metals and plastics should be material-recycled, that there was a near-indifference between incineration and material-recycling of paper and cardboard, and that food waste could equally well be treated by digestion with biogas utilisation or by incineration (Raadal et al., 2009).

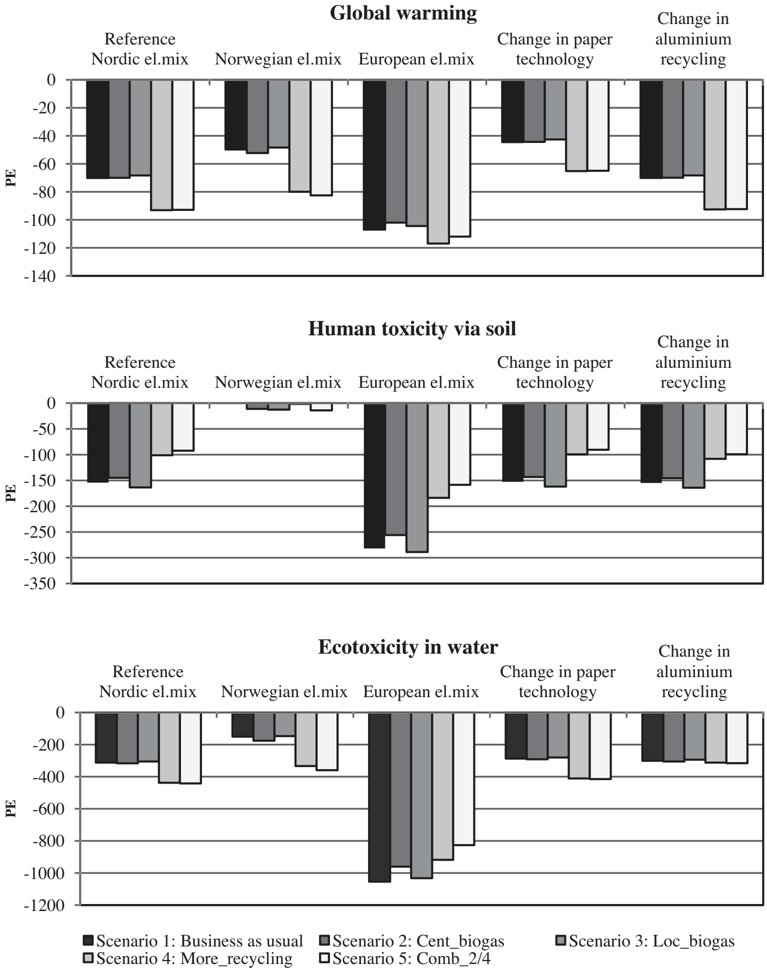

Fig. 5. Sensitivity analysis. Comparing changes in electricity mix, and paper and aluminium technology to the baseline scenarios. It is important to notice the difference in scale between the diagrams.

In a study of different waste treatment options for three Swedish municipalities, it was found that the differences among material recycling, nutrient recycling and incineration were small, but to some extent, the recycling of plastics was better than incineration, while biogas production was worse (Eriksson et al., 2005). Tyskeng and Finnveden (2010) reviewed research done on different waste fractions, and found that for the paper, plastic, metal and glass fractions of the wastes, recycling was, in general, somewhat better than incineration. Local conditions must, however, always be considered, and for the Brøset case we found paper and cardboard recycling to be the most important contributor to savings in Global warming impact. Scenario 4 with increased recycling of paper and cardboard were therefore beneficial for the system concerning this impact category. The difference between recycling and incineration was, however, small, the same as concluded in the studies above. In contradiction to all the above studies where recycling of plastic is somewhat better than incineration, there was no appreciable difference between improved source separation of plastic and incineration in the Brøset case. In the system analyses of the Norwegian waste management system, material recycling of plastic waste resulted in savings to the environment while incineration resulted in an environmental burden. There could be many reasons for this difference in results. Compared to the Norwegian study, the Brøset study uses a higher utilisation factor, with 86% efficiency of the incinerator and 90% utilisation (10% loss in the distribution net) of the energy produces from the incinerator, compared to 85% efficiency and 75% utilisation of the energy as an average for Norway. In addition, there are differences in impact from the replaced energy and the recycling processes. Plastic waste constitutes only 11% of the waste at Brøset based on mass, and even if the recycling benefits should be underestimated, the potential for large savings is small for this fraction.

Metal recycling is beneficial in all impact categories, and an effort should therefore be put into increasing the recycling rates on paper and metal recycling especially. We do not intend to discuss how increased recycling could be achieved, except for one obvious thing to consider, availability of recycling points. Glass and metal recycling containers are usually placed further away from the households as compared to recycling points for fractions like paper and plastic. If the degree of metal recycling should be increased, accessibility is important.

Even if the settlement under planning has carbon–neutrality as its main goal, this study concerns more impact categories than just Global warming, to deal with trade-offs in the system. For both increased paper and metal source separation such trade-offs are present. By increasing the source separation as in scenario 4, the savings in Human toxicity via soil will be reduced, the same holds for the resource Nickel. There are also other practical, economic and social concerns that have to be taken into consideration; we are not discussing these in the article.

We saw from the calculations of waste prevention that less waste entering the system gave less savings in some of the important impact categories, the magnitude of this change was dependent on the waste composition. Gentil et al. (2011) showed how waste prevention can reduce the impact from the production of goods, and how this is more important than the changes in the waste management system. Food waste prevention, especially avoided meat production, was of special importance. While the approach used by Gentil et al. (2011) can be used for some fractions, a general reduction in production due to waste prevention is more problematic to estimate with the use of LCA due to the large amounts of products that would need to be included in the calculations. Hybrid-LCA models could be a solution. We do not quantify the potential for savings in impact due to reduced production of goods in this article, but there is a potential for waste prevention in a settlement like Brøset and this will be investigated further as the planning of the project proceeds. Some actions are suggested by Cox et al. (2010) to promote waste prevention; this is to decide on prevention targets, producer responsibility, householder charging (differentiated waste fees for different fractions), funding for pilot projects, collaboration between the public, private and third-parties, and public intervention campaigns, focusing on special fractions.

Introducing biological treatment of food waste is indifferent to treating this fraction in the incinerator. The results are in accordance with the study of the Norwegian waste management system (Raadal et al., 2009). This means that introducing source separation of food waste in the entire city or locally at Brøset not can be argued in the impact from the waste management system. There could, however, be other reasons for introducing source separation of this fraction, like concern about resource depletion of phosphorous or the importance of co-digestion with other waste sources. The results are, of course, sensitive to the efficiency of biogas production which can be challenging, at least in small-scale systems.

Being a study in the early-stage planning phase, uncertainty is unavoidable; this concerns waste composition, sorting efficiencies, technology both in the present system and in alternative systems, and the potential for increased recycling. Still some processes and parameters are recognised as more important than others. We saw from the sensitivity analysis how changes in the energy mix changed the total loads or savings, and even changed the ranking of the scenarios in some impact categories. A study by Gentil et al. (2009) showed that for countries with high levels of materials recycling and energy recovery, and with an electricity mix causing high GHG-emissions, the waste management system has a potential for savings in emissions. The results were sensitive to changes in the electricity mix, and the assumption done on this parameter is therefore important for the outcome of the study. We have the same results for Brøset where the savings from the system at Brøset are very small if we use the Norwegian electricity mix. We still defend the use of the Nordic electricity mix, because Norway is part of the Nordic electricity market and a net importer of energy.

The efficiency of the incinerator plays an important role due to the large share of waste being incinerated and because of the importance of substituted energy. Brøset as a new settlement cannot affect the efficiency of the incinerator; still it is important to recognise the importance of process optimisation. This concerns the efficiency of hot water production, use of alternative heating sources to cover top load in the district heating system, emission control and the types of energy the district heating system replaces. The incinerator in Trondheim only produces heat; this affects the efficiency of the plant. Co-generation plants have potential for higher efficiencies, and high efficiencies together with improved flue-gas cleaning has been the objective for waste incinerator plants for many years, both due to regulations and for economic reasons (Damgaard et al., 2010).

6. Conclusions

The use of industrial ecology tools in the early stage planning phase can aid in the understanding of what the most important contributors to the environmental impact of a system are. We have used a combination of scenario building and LCA and the results show that for this new settlement located in a city with high recycling and energy recovery rates, introducing source separation and digestion of organic waste is indifferent to the present system, while increasing the source separation of paper and metal would have some benefits. Today's system has savings in Global warming impact and can safely be implemented in the new carbon–neutral settlement. We conclude that at the same time as it is important to perform an assessment of the waste handling system in the early stage planning phase, the research focus in the future should be on waste prevention, where we have less knowledge about drivers and successful instruments, and where the potential for savings could be increased due to saved impact from production of goods. Waste prevention has few, if any, success stories of extensive waste prevention at an acceptable socio-economic cost, and there is little literature on this issue in general. This new settlement should therefore focus on increasing paper and metal source-separation rates, but what should also be emphasised upon is better understanding of the waste prevention issue.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2012.03.018.

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Supplementary data
LCA for household waste management when planning a new urban settlement
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Waste composition

Waste composition is estimated based on analyses of the waste fractions in different areas in Trondheim, and adjusted to the waste fractions used in Easewaste (Kirkeby et al., 2007). There are 48 waste categories in Easewaste and some miss-sorting is included, meaning that the source separated paper fractions can be contaminated with for example plastic and textiles. The degree of miss-sorting is, however, assumed to be low.

Energy mix

All the processes use the Nordic electricity mix (NORDEL), retrieved from Simapro (Pré Consultants, 2011) and adjusted to Easewaste. In the sensitivity analyses the electricity mix is changed to Norwegian electricity and European electricity mix based on the available processes in Simapro.

Transportation

All transportation is included. Brøset is 12 km from the incinerator, the planned digester, and the MRFs for paper and plastic. The MRF for glass and metals is 600 km from Brøset. Plastic is sent by trains to Germany, a distance of 1500 km.

Incineration

We have used a grate furnance Easewaste process and adjusted it with the present energy use, emission factors and handling of ash and aluminium. The incinerator in Trondheim has an efficiency of 86 %. In addition there is a 10 % loss in the distribution net.

Paper recycling

Of the source separated paper 67.5 % is sorted as D-ink and goes to newspaper production; 16.5 % is sorted as cardboard and used for test-liner production; 14 % is sorted as drinking and food cardboard and sent for recycling in Europe where it is assumed to replace printing paper production; and a small fraction (2 %) is sent for incineration with heat recovery. Currently, both D-ink and cardboard are recycled in Norway, so the distances to the recycling plants are short. The Easewaste processes used are Nortern-Europe technology, and all substituted paper is based on virgin production. The substitution and avoided production ratio are between 84 and 100 %.

Plastic recycling

Hard and soft plastic, which together comprise 53 % of the total is granulated and used for new plastic products, while 47 % is sorted as lower-quality plastic, with some granulated and used for products substitution wood, and some used as fuel in the cement industry. The residues (4 %) resulting from the sorting process are incinerated with heat recovery (not shown in Figure 1 in the article). In scenario 4 and 5 we assume higher quality source separated plastic with 65 % going to new plastic products, 31 % substituting wood and 4 % used as fuel.

Glass and metal recycling
Source-separated glass is substituting virgin glass production, with a loss of 11 % in the process. Aluminium is substituting virgin aluminium production, with a loss of 11 % in the process. Steel is substituting steel production mainly based on virgin Iron (89 %) with a substitution ratio of 100 %.

Digestion
We have used a Easewaste process for biogas production and adjusted it for energy use of a small-scale and a large-scale digester according to Berglund and Börjesson (2006). The methane content of the biogas is assumed to be 62 % with a 2 % loss of methane in the process. The digestate is used on land close to the digester and avoids production of K, N and P fertilizers. In scenario 2 the biogas is upgraded to fuel, and substitutes diesel used in buses, while in scenario 3 we assume the biogas to be used in a CHP-plant, with 50 % heat production and 30 % electricity production.

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Paper 2

Influence of assumptions about household waste composition in waste management LCAs.

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Influence of assumptions about household waste composition in waste management LCAs

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ABSTRACT

This article takes a detailed look at an uncertainty factor in waste management LCA that has not been widely discussed previously, namely the uncertainty in waste composition. Waste composition is influenced by many factors; it can vary from year to year, seasonally, and with location, for example. The data publicly available at a municipal level can be highly aggregated and sometimes incomplete, and performing composition analysis is technically challenging. Uncertainty is therefore always present in waste composition. This article performs uncertainty analysis on a systematically modified waste composition using a constructed waste management system. In addition the environmental impacts of several waste management strategies are compared when applied to five different cities. We thus discuss the effect of uncertainty in both accounting LCA and comparative LCA. We found the waste composition to be important for the total environmental impact of the system, especially for the global warming, nutrient enrichment and human toxicity via water impact categories.

1. Introduction

Life cycle assessment (LCA) has been used for many years in waste management research in order to estimate the life cycle impact of waste management and to compare and optimise systems (Bergsdal et al., 2005; Christensen et al., 2009; Iriarte et al., 2009). The ultimate goal for an LCA is to model the real world system as accurately as possible. Waste management systems are, however, complex, and uncertainty in the modelling is unavoidable. Managing these uncertainties therefore becomes important. Gentil et al. (2010) made a qualitative comparison of different waste LCA-models and found the functional unit, system boundaries, waste composition and energy modelling to have a potentially significant impact on the results. Large uncertainties were also found by Winkler and Bilitewski (2007) when applying identical system assumptions in different LCA-tools. Merrild et al. (2008) found that both technology choices and system boundaries have a significant impact on the results when comparing recycling and incineration of paper waste, and Rigamonti et al. (2009) discussed the importance of choices in material and energy recovery parameters when assessing integrated waste management systems. While the importance of technology choices, system boundaries and energy substitution are discussed in the literature, much less attention is paid to uncertainty in the waste composition. Christensen et al. (2009) included a variation in waste composition when performing sensitivity analyses on the assessment of 40 generic waste management scenarios. However, only the global warming impact category was included in this study.

The composition of waste influences the potential for recycling, substitution of other heat and/or electricity sources, and biogas production; it also influences the environmental impact from incinerators and landfill. Waste composition is therefore potentially very important for the outcome of LCAs of waste management systems. Waste composition is defined by the weight distribution between different fractions and the chemical composition of each fraction. Uncertainty in both of these parameters can influence the results. Riber et al. (2009) used two methods to decide the chemical compositions of 48 material fractions in Danish household waste, which are included in the LCA-waste tool EASEWASTE (Kirkeby et al., 2006). We will, however, not discuss uncertainty in chemical composition in this article, but concentrate instead on uncertainty in the weight distribution between different fractions.

Waste compositions are not usually described to the level of detail found in the EASEWASTE software. Assessments are usually based on municipality or company specific data, national average data, or waste composition analysis. In Norway most municipalities have recycling programmes, and the size of each sorted fraction is reported annually. Dahlen et al. (2009) discussed the many sources of uncertainty for such public data. They identified sixteen sources of error and uncertainty in the interpretation of official waste collection data in Sweden, and sorted them into four main categories: general data problems, data uncertainties related

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to specific waste categories, unreliable data from recycling centres and non-comparable household-waste composition analysis. The large variation in how waste composition analysis is performed proved challenging when an average waste composition for Norwegian household waste was estimated in 2010 (Skullerud et al., 2010). Analysis from 33 of 85 municipalities had to be rejected because of data quality problems. Problems with composition analysis include inconsistency in the classification of fractions (Dahlén and Lagerkvist, 2008) and levels of aggregation (Riber et al., 2009). A high level of aggregation often results in a large 'other' fraction, which can be above 20% in some studies (Gentil et al., 2009; Skullerud et al., 2010), this introduces additional uncertainties to overall chemical composition of the waste. Highly aggregated fractions also make it challenging to distinguish between different qualities of recyclables. Paper and cardboard collected together, for example, are usually sorted in different fractions with different pathways in the recycling system, and there are differences in the impact from recycling high quality paper and cardboard to those arising from recycling low quality materials.

In order to contribute to the discussion of uncertainty in LCAs performed on waste management systems, the objective of this article is to estimate the impact of uncertainty in the waste composition of household waste when applying the same LCA-tool, functional unit, technologies and system boundaries in all scenarios. The waste management system is modelled to reflect a typical Norwegian city. In addition uncertainty in comparative assessments is investigated by using a range of source-separation scenarios.

2. Methodology

This study investigated how changes in assumptions regarding waste composition affect the modelled environmental impact of a waste management system. This was achieved by first identifying a representative waste management system for Norwegian conditions. Then an average waste composition was estimated, using a fairly low level of aggregation. Next we decided how the waste composition should be systematically modified in the assessment and which sorting efficiencies should be used. In conclusion we looked at how environmental impact was affected by changes in sorting efficiencies and technology in an assessment of five cities with different waste compositions.

2.1. Waste management system

A hypothetical Norwegian household waste management system was modelled. In 2011 56% of Norwegian household waste was incinerated, 40% was collected for recycling, 2% was landfilled and 2% received other treatment (SSB, 2012a). Landfilling of organic waste was forbidden in 2009. Most municipalities collect from households using a two- or three-bin, or container, system. In addition most municipalities have neighbourhood collection points, where household deliver glass and metal waste for recycling. Household source-separated waste such as wood, garden waste, EE-waste and hazardous waste has to be brought to recycling centres (EE-waste can also be delivered at stores retailing EE-products). The uncertainty analysis focused on the fractions collected from households and delivered to glass/metal recycling, while source-separated garden waste, textiles, EE-waste and wood were left out of the calculations, as was bulky waste delivered to the recycling centres. We assumed a three-bin system in addition to collection points for glass and metals. The waste was assumed source-separated into the following fractions: (1) mixed waste, (2) paper and cardboard, (3) plastic, and (4) glass and metals (Fig. 1). Sorting of food waste has been implemented in only some cities in Norway so far, and was not included in the assessments of the waste system. However, we included food waste sorting in the comparative assessment of alternative scenarios performed subsequently. All the source-separated recyclables were sent to Material Recovery Facilities (MRFs). Paper was sorted in three fractions plus residues, plastic in two fractions plus residues, and glass and metals in three fractions without residues. All the waste fractions were followed to their end destination, meaning that avoided environmental impact from substitution of virgin materials was included. Mixed waste was incinerated with heat recovery. The destination of the incinerator residues, such as fly ash and bottom ash, was not included; neither was metal extraction from the bottom ash.

Transportation was included in the model, but with fixed distances of 20 km for transport from collection to the sorting facilities or the incinerator, and of 100 km for transport from the sorting facilities to the recycling centres. The collection vehicles were assumed to be the same for all fractions. Similarly it was assumed that the same type of truck was used for the long distance transport of all fractions. This approximation was used in order to make the results as generally applicable as possible. In any case, transportation has been shown to have very little influence on the performance of waste systems as long as the waste is not transported for very long distances (Salhofer et al., 2007). The Nordic electricity mix was used as the energy source for all processes taking place in Norway. For processes carried out in Europe, such as plastic recycling, an EU-mix was applied. For this we used an average approach, assuming that changes in the waste system will not affect the use of marginal energy in the countries involved. The approach taken to energy in waste management LCAs is subject to on-going discussion. While the ILCD Handbook (EU JRC, 2010) recommends the use of average technology assumptions when performing attributional LCA, and the use of marginal technology assumptions for consequential LCA only, no recommendations are widely agreed upon or followed (Bernstad et al., 2011; Fruergaard et al., 2009). Contribution to this discussion is not included in the scope of this study.

2.2. Waste composition

In Norway there are public reporting systems for household waste, whereby municipalities report the quantities of waste they recycle, incinerate and landfill annually. Although source-separated waste is reported by fraction, the data still do not provide enough information with which to estimate the composition of household waste. In order to establish a reference scenario for waste composition with a sufficiently detailed aggregation level, we chose to combine the data reported by the municipalities of five cities (SSB, 2012b) with composition analyses from the same five cities (NORSAS, 2007; RiG and Asplan Viak, 2011; RKR, 2007). The reference waste composition used was therefore not a Norwegian average, but represented an average of these five cities. The cities vary in size between 5000 and 170,000 inhabitants, and therefore represent small to large Norwegian cities. The waste fractions were aggregated into nine main fractions and then an average of each fraction for the five cities was calculated (Table 1). This average waste composition is referred to as the reference scenario. In order to increase the level of detail, the 9 main fractions were then disaggregated into 18 fractions. The disaggregation was based on data available in the aforementioned composition surveys, a Norwegian report on the waste management system of Norway (Raadal et al., 2009) and the average waste composition for Danish households in 2005, found in EASEWASTE. Paper and cardboard was disaggregated into four fractions (newsprint 65%, paper and cardboard containers 13%, milk cartons 7% and other paper 15%); plastic was disaggregated into three fractions (soft plastic 50%, hard plastic 30% and non-recyclable plastic 20%); food was disag-

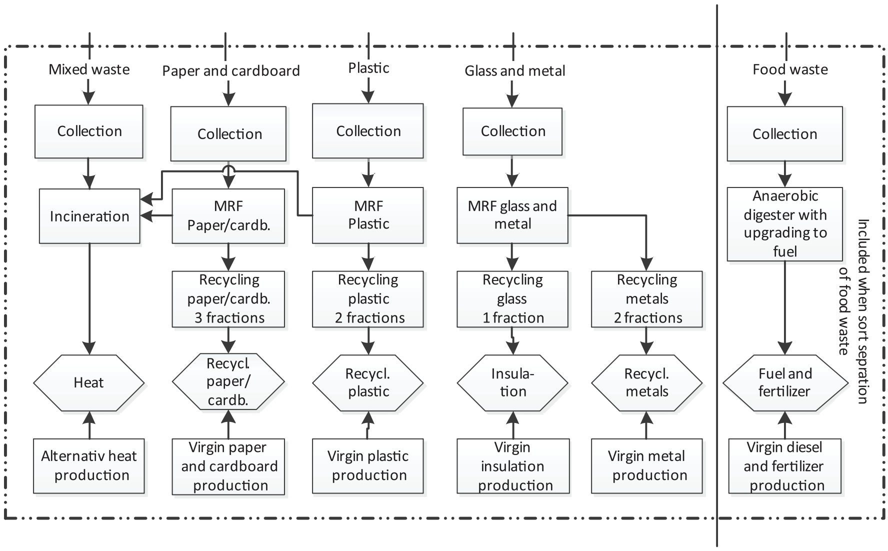

Fig. 1. The modelled waste management system.

Table 1
Waste composition of five Norwegian cities and the calculated reference scenario.

Fraction	Trondheim[a]	Kristiansand[b]	Arendal[b]	Sogndalen[b]	Skien[c]	Reference scenario
Paper and cardboard	37.0	28.9	33.7	28.7	29.2	31.5
Plastic	12.1	15.2	13.1	17.4	8.2	13.2
Food	24.8	27.9	23.6	24.2	28.7	25.8
Glass	5.7	4.1	6.1	5.2	5.4	5.3
Metal	5.0	4.6	5.2	6.2	6.6	5.5
Garden waste	2.2	6.4	1.4	3.5	2.0	3.1
Textiles	3.4	4.1	3.4	4.4	2.6	3.6
Other combustible	5.1	6.5	11.6	8.2	9.5	8.2
Other non-combustible	4.7	2.3	1.9	2.1	7.9	3.8

[a] NORSAS (2007) and SSB (2012a,b).
[b] RKR (2007) and SSB (2012a,b).
[c] RiG and Asplan Viak (2011) and SSB (2012a,b).

gregated into two fractions (vegetable food 76% and animal food 24%); glass was disaggregated into two fractions (recyclable glass 96% and non-recyclable glass 4%) and metals into three fractions (aluminium 10%, steel 70% and other metals 20%). We do not discuss the uncertainty within these fractions in this article. There are, however, differences between the data from the cities, the data from Denmark, and the report from the Norwegian system in terms of both the numbers of disaggregated fractions and the distribution between them. This is, however, of little importance for the objective of this study.

Large variations in the weight-percentages of the main fractions in the five cities were found. To perform a systematic analysis of the consequences of uncertainty, the main fractions (paper, plastic, food, glass and metals) were in turn increased or decreased by 15%. The remaining fractions were adjusted afterwards, based on their percentage by weight. The reference waste composition and the 10 alternative waste compositions are found in Table 2.

2.3. Sorting efficiencies

The total impact from the system was calculated based on the quantities of each recycled fraction and the quantity and waste composition of the incinerated waste. The quantities of waste recycled and incinerated can either be calculated using estimated source-separation efficiencies or taken from measured quantities of these fractions. Both cases were modelled in this study (Fig. 2).

In the first case (Case 1) we kept the sorting efficiencies constant and at a level based on the typical sorting efficiencies found in many Norwegian cities. A change in waste composition combined with constant sorting efficiency resulted in changes in both the quantities of recyclables and the quantity and composition of waste incinerated. This calculation method relates to uncertainty in waste composition when performing LCAs on new systems or when introducing changes to existing systems, such as increased sorting efficiencies or the introduction of food waste sorting. The sorting efficiencies were set to 75% for paper and cardboard, 25% for plastic, 60% for glass and 65% for metals. In addition we assumed that 10% of the non-recyclable paper and cardboard was sorted as paper and that 10% of the non-recyclable plastic was sorted as plastic. These sorted non-recyclable fractions were separated at the MRFs and sent for incineration.

In the second case (Case 2) the quantities recycled and incinerated were kept constant. A variation in waste composition will in this case change the sorting efficiencies used in the calculations. This method is used for accounting LCA in existing systems when the waste composition and quantities of waste are known. Case 2

Table 2
Waste composition of the calculated reference scenario and the 10 alternative scenarios.

Fraction	Ref.	Paper high	Paper low	Plastic high	Plastic low	Food high	Food low	Glass high	Glass low	Metal high	Metal low
Paper and cardboard	31.5	36.2	26.8	30.8	32.2	29.9	33.1	31.2	31.8	31.2	31.8
Plastic	13.2	12.3	14.1	15.2	11.2	12.5	13.9	13.1	13.3	13.1	13.3
Food	25.8	24.1	27.6	25.3	26.4	29.7	22.0	25.6	26.1	25.6	26.1
Glass	5.3	4.9	5.7	5.2	5.4	5.0	5.6	6.1	4.5	5.3	5.4
Metal	5.5	5.1	5.9	5.4	5.6	5.2	5.8	5.5	5.6	6.3	4.7
Garden waste	3.1	2.9	3.3	3.0	3.2	2.9	3.3	3.1	3.1	3.1	3.1
Textiles	3.6	3.3	3.8	3.5	3.6	3.4	3.8	3.5	3.6	3.5	3.6
Other combustible	8.2	7.6	8.7	8.0	8.4	7.8	8.6	8.1	8.3	8.1	8.3
Other non-combustible	3.8	3.5	4.0	3.7	3.9	3.6	4.0	3.8	3.8	3.8	3.8

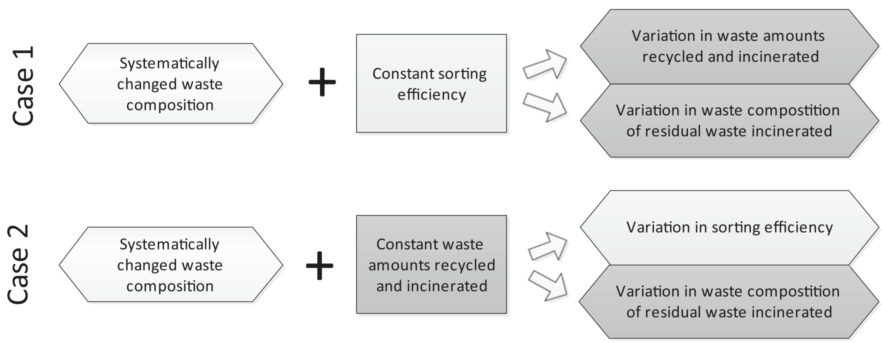

Fig. 2. Two methods estimating the consequences of changes in the waste composition. Case 1 is for new systems, or systems with changes in technology, while Case 2 is for existing systems.

Table 3
Alternative sorting efficiencies used in the comparative LCA.

Fraction	Reference sorting efficiency %	Increased paper source sep. %	Increased plastic source sep. %	Increased metal source sep. %	Food waste sorting %	All measures %
Paper and cardboard	75	90	75	75	75	90
Plastic	25	25	50	25	25	50
Food	0	0	0	0	50	50
Glass	60	60	60	60	60	60
Metal	65	65	65	90	65	90

did not include uncertainty in how much waste is sorted or the quality of this waste. Changes in waste composition therefore only influenced the impact from the incinerator and the avoided impact from substituted energy. The functional unit, the system boundaries and the technologies remained constant.

One common reason for applying LCA to waste management systems is to compare alternative solutions for waste management. We wanted to analyse the robustness of scenario comparison to uncertainty in the waste composition. We used the waste composition in the five cities included in the study, with the same disaggregation factors used for the reference scenario. In five alternative scenarios we first applied an increase in source separation of paper and cardboard from 75% to 90%, secondly an increase of plastic source separation from 25% to 50%, thirdly an increase of metal recycling from 65% to 90%. Fourthly, we introduced food waste sorting and biogas production with a 50% sorting efficiency, and finally we included all the measures in the first four scenarios together (Table 3). Since the sorting efficiencies were given, the Case 1 calculation method was used.

This method gave us three types of results:

- the change in impact in the presence of uncertainty in the waste composition with assumed constant source-separation efficiency (Case 1);
- the change in impact in the presence of uncertainty in the waste composition with assumed constant quantities of source-separated fractions (Case 2);
- the effect of uncertainty in waste composition when comparing alternative waste strategies.

2.4. Impact assessment

All the modelling was performed in the 2012 version of the EASEWASTE LCA-tool (Kirkeby et al., 2006). EASEWASTE is a product developed by the Technical University of Denmark and is designed for waste management LCA. It uses the EDIP method as the default mid-point impact assessment method (Wenzel et al., 1997). Characterisation values were updated in 2004. The impact categories calculated with EDIP are resource depletion, human toxicity via water, air and soil, ecotoxicity in soil and water, stratospheric ozone depletion, acidification, photochemical ozone formation, and nutrient enrichment. The normalisation values can be found in Laurent et al. (2011). For global warming the IPCC, 2007 characterisation factors were used (IPCC, 2007). To calculate the environmental impact from the waste system we used a functional unit consisting of the treatment of one tonne of household waste.

3. Results

From the calculation methods used we obtain three types of results: the first and second from the two methods for estimating total impact (Cases 1 and 2), and the third from the comparison of alternative waste management systems. Impact categories contributing to savings less than 0.5% of one person's annual average impact are not included in the results. This leaves us with six impact categories, of which human toxicity via water is the most important due to replacement of virgin aluminium production.

3.1. Cases 1 and 2

Results from the systematic ±15% changes in the paper, plastic, food, glass and metal content of the waste are shown in Fig. 3. All results are given as the percentage change in total impact compared to the total impact when applying the reference waste composition. Positive percentage changes in the figure indicate an improvement in impact.

The *resource depletion* impact category is an aggregation of the use of fossil fuels, metals and renewable resources. Use of gravel, sand, clay and limestone are not included. There are two important processes for the resource depletion impact category in this study: replacement of electricity production by the heat produced in the incinerator and the substitution of virgin aluminium production. The total resource depletion impact for the reference scenario is negative, indicating a net saving in resources. As we can see in Fig. 3, the scenarios with large amounts of waste with high heating value going to the incinerator and increased weight of metals to recycling will improve the system (paper low, plastic high, plastic low, food low, glass high and metal high). With a ±15% change in each component of the waste, the variation in outcome in Case 1 (constant sorting efficiency) is between 10% more and 3% less savings than the reference scenario, and in Case 2 (varying sorting efficiency) between 3% more and 4% less savings. In the Case 2 scenarios the quantities of waste going to the incinerator and the source-separated metals are known, and changes in waste composition is therefore of less importance to the total impact from the system.

The *global warming* category aggregates the greenhouse gas emissions into CO_2-equivalents in a 100 year perspective. For the reference scenario the total saved impact on global warming was

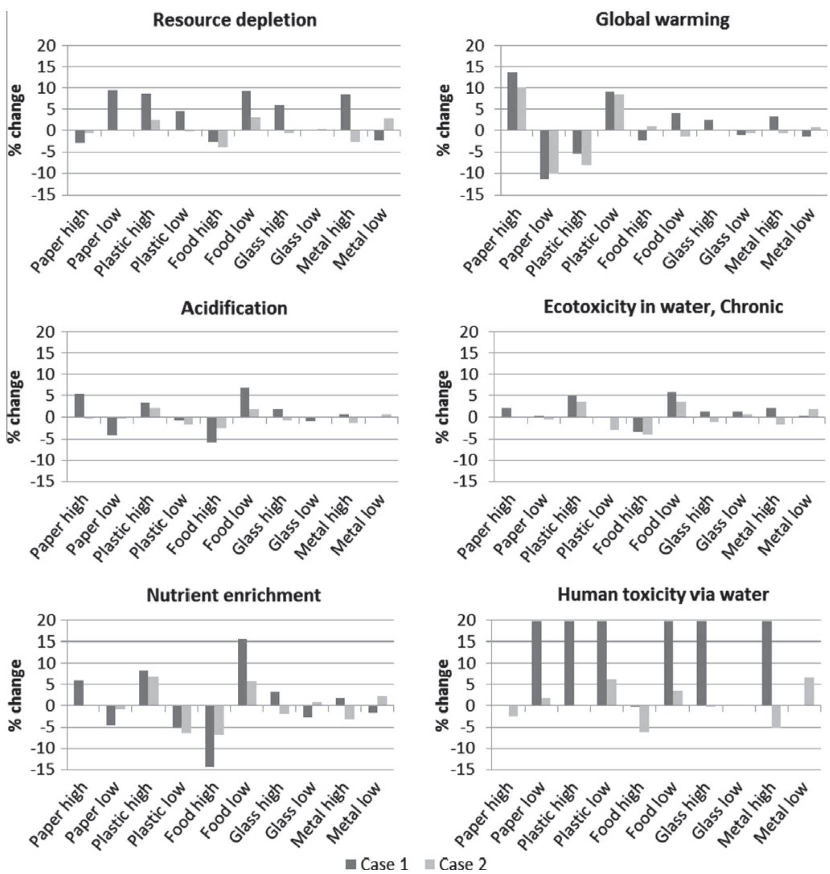

Fig. 3. Changes in total impact related to the reference scenario. Increased savings in impact will give a positive bar in the figure.

found to be 322 kg CO_2-eq. per tonne of managed waste. In Fig. 3 we can see how a ±15% change in each component of the waste gives a variation in outcome for Case 1 between 14% more and 12% less savings than the reference scenario, and in Case 2 between 10% more and 10% less savings. Uncertainty in waste composition has therefore a relatively large influence on the total impact from the system in this category. The most important fractions for both Case 1 and 2 are paper and plastic. A large paper fraction is important for both the heating value of the waste to the incinerator and the favourable replacement of virgin paper production. While the CO_2-emissions from burning paper are counted as carbon neutral, the plastic is a non-renewable resource and the CO_2-emissions are accounted for in the total impact. More paper in favour of less plastic is therefore positive for the total savings in impact in this category.

The *acidification* category aggregates all emissions leading to acidification into SO_2-equivalents. From Fig. 3 we can see that a ±15% change in each fraction of the waste give a variation in outcome for Case 1 of between 7% more and 6% less savings than in the reference scenario, and in Case 2 of between 2% more and 2% less savings. Changes in the percentage weight of paper and food are the most important. This is because these two fractions are the largest, and the amount of replaced virgin newspaper production is the most important process for increasing the savings in impact in this category. When the quantity of source-separated material is known (Case 2) there are small uncertainties connected to this category.

The *ecotoxicity in water* category aggregates all toxic emissions potentially impacting the environment into units of m^3 water. With a ±15% change in each component of the waste, the variation in outcome for Case 1 is between 6% more and 3% less saving, and in Case 2 between 4% more and 4% less savings than the reference scenario. The replacement of electricity from the heat produced in the incinerator is the main contributor to the saved impact. The changes in impact are therefore relative small for all scenarios in both cases.

The *nutrient enrichment* category aggregates all nutrient-enriching emissions into NO_3-equivalents. In this category the two most important processes are replacement of virgin newspaper production and emissions from the incineration process due to the food content of the waste. In Fig. 3 we can see that with a ±15% change in each component of the waste, the variation in outcome for Case 1 is between 16% more and 14% less savings, and in Case 2 between 6% more and 7% less savings than the reference scenario. Increasing

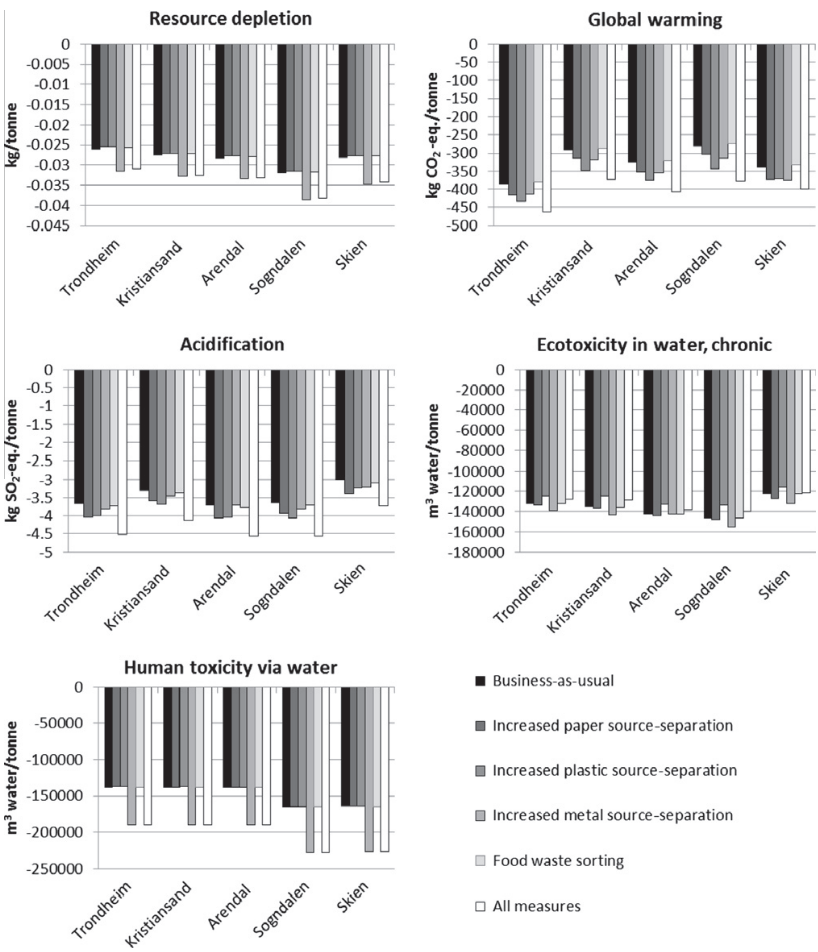

Fig. 4. Total impact for scenarios applied to each city.

the food content of the waste has the largest effect on this category because we get a combination of more impact from the incinerator due to more NO_x-forming waste and because a 15% change in food content has a large impact on how much paper is recycled. The nutrient enrichment category should, given this large percentage change in impact, be seen as very sensitive to the percentage weight of food waste. However, the importance of this category for the total environmental impact of the waste management system, after normalisation, is low.

The *human toxicity* category aggregates all toxic emissions potentially affecting human health into units of m^3 water. This impact category contains the largest savings, after normalisation. The reduction in impact is due to substitution of virgin aluminium production. Fig. 3 shows how a ±15% change in each component of the waste in Case 1 will either give a result equal to the impact of the reference composition or provide a 20% increase in savings. The 20% increase in total avoided impact is due to a change in aluminium content from 0.5% to 0.6%. The human toxicity category is therefore very sensitive to the content of aluminium in the waste, and most importantly to how much of the available aluminium is recycled. The changes in total impact are much smaller for Case 2. This is because the quantity of aluminium recycled is approximately the same in all scenarios. However, due to the level of accuracy in the model used, there are some small differences in the quantity of aluminium recycled, even in Case 2. Although these differences are less than 10 kg recycled aluminium per tonne total waste, they are nevertheless the reason for the variation in the Case 2 results.

3.2. Alternative scenarios

In order to examine the importance of waste composition for the results of comparative assessments we compared scenarios for the waste management system applied to five cities (the same cities used for estimation of the reference waste composition). All impact categories have a net negative impact, indicating avoided impact from the modelled system (Fig. 4). We can see that there are relative large differences between the cities in some impact categories.

When we increase the source-separation efficiencies and introduce sorting of food waste we can see that the results are robust for the resource depletion and human toxicity via soil impact categories. In these impact categories the prioritisation of measures is the same for all cities involved, with increased metal source-separation as the most important measure. For global warming, acidification and ecotoxicity in water, the ranking between measures can be altered based on waste composition. For global warming all scenarios increasing the source-separation of paper, plastic and metals increase the avoided impact compared to business-as-usual. It is the ranking between the fractions that depends on the waste composition. Ecotoxicity in water is the only impact category where introducing all measures has a negative effect on the total impact for all cities. The reason for this reduction in savings from the systems is the increased source-separation of plastic waste which reduces replaced energy from the incinerator.

4. Discussion

We wanted to study the effect of uncertainty in waste composition in a waste management system comprising the recycling of paper, plastic, glass and metals, and the incineration of residues with heat recovery. We have combined waste composition analyses and the annual amounts of source-separated waste fractions reported by the relevant municipal authorities to estimate waste compositions for five cities. There can be many reasons for uncertainty in the estimated waste composition, stemming from uncertainty in both the composition analyses and in reported waste amounts, and from the combination of these two sources of information. A Norwegian study compared the composition analysis of 52 municipalities and found an uncertainty in each waste fraction of between three and six per cent (hazardous waste and textiles had higher uncertainty) (Skullerud et al., 2010). Gentil et al. (2009) found large variation in literature data on waste compositions when performing an assessment of the waste systems in several European countries, and Dahlen et al. (2009) discuss uncertainty in public waste data. Large difference can be found if we compare the five municipalities included in the present study. Sogndalen has a paper fraction with a percentage by weight of 28.7, while the paper percentage by weight for Trondheim is 37. For plastic the percentage by weight is 8.2 in Skien, while in Sogndalen it is 17.4. The reason for these differences could be due to the different sizes of the cities, their geographical location, seasonal variations influencing waste composition analysis, etc. A comparison of waste compositions in Norwegian cities found large cities to have a larger percentage by weight of paper, hazardous waste and other non-burnable waste, and less metal, food and other burnable waste (Skullerud et al., 2010). We will, however, not go further into the reasons for uncertainty in waste composition, but rather discuss the effect of such uncertainty on modelling of waste systems.

The paper fraction is the largest fraction in all cities. This fraction is important for the global warming impact category due to the avoided impact of paper production when recycled and the heating value when incinerated. We found that changes in paper content had the largest effect on global warming impact and that the cities with the largest paper fractions (Trondheim, Arendal and Skien) had the most avoided impact. There is, however, an additional reason for these results. When there is a large paper fraction, the plastic content of the waste is low. Increased plastic content has the opposite effect on the global warming category; it will decrease the avoided impact. A 15% change in paper waste will give a larger effect on the total waste composition than a 15% change in plastic content. When we, however, compared the different increased recycling rates, we found that increasing the plastic source-separation from 25% to 50% would be more important than increasing the paper source-separation from 75% to 90%. One exception is the town of Skien where the plastic content of the waste is very low. When we systematically modified the waste composition the largest change in global warming, 43 kg CO_2-eq per tonne, was found when we had a large paper fraction (and thereby low plastic fraction) compared to the average waste composition. A comparison of the impact from the five cities suggested that the difference in impact could be up to 105 kg CO_2-eq per tonne waste. Christensen et al. (2009) found a difference in global warming impact in the order of 100–200 kg CO_2-eq per tonne waste when applying an average EU composition, a typical 'northern European' waste composition and a typical 'southern European' waste composition, to 40 generic waste management scenarios. The present study confirms the importance of using a representative waste composition when estimating the global warming impact of waste management systems.

Recycling of metals, especially aluminium, proved to be very important for impact categories such as resource depletion and human toxicity via water. The cities with the largest metal content have the largest avoided impact, especially for the human toxicity category, and increased source-separation of metals is important in all cities for resource depletion, human toxicity via water, and even for global warming. The human toxicity via water category is very sensitive to changes in metal content, especially aluminium content. In Norway glass and metals are collected together and separated in a centralised plant serving the entire country. The actual fractions of glass and metal, and of different types of metal, are reported at a national level only. This leaves large uncertainties in assessments at a municipality or settlement level, which will affect the results of an accounting LCA where we have an estimated source-separation ratio. However, uncertainty in waste composition concerning metals is of little importance if the source-separated amounts are known.

Food waste is the next largest fraction after paper for all cities. We found the nutrient enrichment impact category to be especially sensitive to changes in this fraction. The challenge with food waste is often the way in which it is reported. In some waste composition analyses food waste is reported together with garden waste in a single organic waste category, while others report the food waste category separately (Bernstad et al., 2011; Gentil et al., 2009). It is important to be aware of this difference, as food waste and garden waste often have different routes in the waste management system, and therefore also different environmental impacts.

While in accounting-LCA we can find relatively large effects of uncertainty in the waste composition, comparisons of the effect of increased source-separation or introduction of food waste sorting provide more robust results. The total impact is then of little importance, it is the difference between the impacts that decides which solutions are the most desirable. Although there were some ranking issues, these were small. This is supported by findings by Eriksson and Baky (2010) and Christensen et al. (2009).

5. Conclusion

The uncertainty in waste management LCA arising from uncertainty in waste composition has been less extensively studied than uncertainty stemming from the choice of system boundaries, technology and energy replacement. In this paper we have systematically modified the average waste composition estimated from the waste composition of five Norwegian cities. We found that a ±15% change in selected fractions resulted in a greater than 10% change in global warming, nutrient enrichment and human toxicity via water impact categories. Hence, such LCA impacts are highly sensitive to uncertainties in waste composition. If the quantities of source-separated material are known the uncertainty is low for most categories, but still 10% for global warming. A percentage change in the large fractions – paper, plastic and food waste – is of most importance, together with changes in the metal content. When comparing scenarios, the results are more robust. The analysis of the five cities showed wide variation in waste composition in the municipalities, and using a waste composition from another city or an estimated average can influence the result of a study. Having good data on the quantity of waste recycled, and the quality of this waste is of importance for the reliability of the results.

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Paper 3

Using IO-LCA to explore how household waste prevention influences economy-wide GHG emissions.

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Using IO-LCA to explore how household waste prevention influences economy-wide GHG emissions

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Abstract
Waste prevention is at the top at the waste hierarchy, still there are few methods available to quantify the environmental benefits of waste prevention in an economy-wide perspective. There are examples on models including avoided upstream impact with the help of life cycle assessments, but only for fractions where the functionality can be substituted or where there is no reduction in consumption. In this article a hybrid-LCA model is used to combine downstream life cycle assessment with upstream environmentally extended input-output analysis. By applying waste prevention assumptions to this model, the total potential benefits of waste prevention can be calculated and the consequences of different rebound effects examined. The model is applied to the waste management system in the city of Trondheim in Norway, with prevention of household food-, textiles- and paper waste. The results demonstrate the large contribution of upstream impact reductions when discussing waste prevention, but also the importance of what kind of assumptions regarding rebound effects are made.

Keywords: consumption, hybrid-LCA, rebound effect, waste prevention

Introduction
According to the Waste Framework Directive (European Commision, 2008), waste prevention is measures taken before a substance, material or product has become waste, in order to reduce: a) the quantity of waste, including the re-use of products or the extension of the life span of products, b) the adverse impacts of the generated waste of the environment and human health, or c) the content of harmful substances in materials and products. Hence, waste prevention can be quantitative (amounts of waste) and qualitative (adverse impacts or harmful substances).

Waste prevention is at the top of the waste management hierarchy, however, few countries (if any) can report successful large-scale implementation of prevention policies. It seems like waste prevention, despite its top priority in policy, is difficult operationalize in practice. Part of the reason is probably that waste prevention – upstream to the point of waste generation – is outside the direct responsibility of key actors in the waste management sector. On the contrary, such actors earn their living from cost-effective management of waste that is already generated, according to numerous specific waste regulations, and municipalities may also find it difficult to influence the amounts of waste arising from households.

There is very limited literature assessing the environmental benefits of waste prevention. Part of the reason is lack of empirical basis, i.e. prevention projects implemented and suitable for in-depth studies. Another reason is the methodological challenge of assessing impacts from waste prevention. First, it is generally difficult to analyse with accuracy something that is not there (waste not present). Second, environmental impact assessment in waste management, which is normally carried out by LCA methods, seldom covers activities and processes *upstream* to the consumer; the boundary is normally the waste system itself. Due to the complexity of production and consumption systems, and the fact that a given waste fraction that could be subject to prevention is normally a mix of numerous discarded product categories, the use of traditional LCA is just not feasible. Ideally, one would need a methodology that could combine the potential upstream and downstream effects of waste prevention measures, using state-of-the-art assessment methods both on the upstream and on the downstream side. Third, such a combined methodology should take into account the so-called rebound effect that are important regarding the economy-wide consequence of a change in consumption (Hertwich, 2005). Since waste prevention most likely would also lead to a reduced purchasing of some goods, the consumer would save money, and some of this saving would probably be re-spent on purchasing other goods. Hence, waste prevention would probably give a shift in consumption, with reduced environmental impacts from what is not consumed, but with added impacts from what is consumed instead. One would have to estimate how this shift in consumption influences the *net benefits* of waste prevention.

This study examines the potential net benefits with respect to greenhouse gas (GHG) emissions from selected household waste prevention opportunities using a hybrid-LCA methodology. This is a case study, referring to waste prevention in the city of Trondheim, Norway, inspired by an urban development plan for 1500 new households located at Brøset in Trondheim, where the aim is to develop a close to zero carbon emission new urban settlement. In a previous study we examined the potential environmental benefits of different waste treatment and recycling strategies at Brøset (Slagstad and Brattebø, 2012), while this study specifically focuses waste prevention opportunities.

With reference to the above-mentioned challenges, we are convinced that a higher priority to waste prevention *in reality* will not happen unless its potential benefits can be better documented and understood in a system-wide perspective. This is the main motivation behind our study. As a starting point we developed the following hypothesis: "Waste prevention is at the top of the waste management hierarchy, and therefore, we expect that household waste prevention will give large environmental benefits both downstream (in the waste management system) and upstream (in the production system). However, due to the rebound effect, most of the benefits will be lost as a result of added environmental impacts from other products and household expenditure shifts." This article is based on the master thesis of Lébre (2012).

Literature

The waste management hierarchy has been implemented in Norway since the early 90's, and the waste management focus has successfully been moved up the hierarchy from landfilling to incineration and recycling. Only 6 % of household waste was landfilled in 2010 (SSB, 2011). It has, however, been much more difficult to deal with waste prevention. Cox et al. (2010) studied the popularity of different waste prevention options, and found donating of goods and charity at the top, followed by small reuse behaviours around the home, and changes in consumption habits as the least popular option. Salhofer et al. (2008) found measures which do not require a reduction in consumption to only have a potential to prevent 1-3 % of the waste. Still, waste prevention targets of 15-20 % can be found in environmentally ambitious settlements, such as Hammarby Sjöstad in Sweden (Finnson, 2006) and Eco-Viikki in Finland (Energie cites and Ademe, 2008). To reach such ambitious targets one has to consider changes in consumption habits.

Wilson et al. (2010) ask for more attention to waste prevention, and claim that costs, climate change and sustainable use of natural resources are its most important drivers. When modelling the impacts from waste, climate change has mainly been related to the downstream waste management, and there is extensive literature concerning the potential environmental impacts from different waste systems, or parts of such systems. Few models, however, include waste prevention in the calculations. This is partly due to the challenge of how to model avoided waste, when only considering the waste management system. Some authors have expanded their systems to overcome this challenge. Cleary (2010) developed Waste-Map LCA, modelling waste prevention in a way where the service level of the related products is maintained. The waste-LCA system boundary was expanded by product-LCA for the substituted waste fractions, assuming that prevented goods would be replaced by other goods, with the same function, but with a lower environmental impact. The study explained how waste reduction is the highest form for waste prevention, however, it was found that without substitution of service level this is also the option that is most likely to fail. Gentil et al. (2011) also used an extended LCA-model, however, they included waste prevention with no direct substitution of functionality, and only for fractions where reduced waste amounts can occur without reduction in consumption, such as for food and unsolicited mail. They found waste prevention to be beneficial, especially with respect to avoided upstream production and when dealing with low-technology waste management systems. There are, however, as far as we know, no models available for analysing waste prevention in the cases where there is a change or shift in consumption not maintaining the functionality of the goods involved.

If waste prevention leads to reduced consumption, households will have extra money available. This can be re-spent on other goods or services, it can be saved, or the household can choose to reduce its income and therefore spend less. The rebound effect relates to the first option. Alfredsson (2004) used an environmentally extended input-output (EIO) model to examine the consequences of 'green' consumption, where she analysed the effect of changed consumption patterns on CO_2-emissions. Her conclusion was that green consumption has limited effect because of the rebound effect. This phenomenon is also discussed by Druckman et al. (2011) who studied why only a portion of the greenhouse gas emission reduction estimate for UK households is achieved in practice.

Food consumption comprises a fairly large share of total environmental impact from households (Hertwich, 2011), and according to Tukker et al. (2010) it should be one of the priority areas for targeting impact from households in general. In England, two different food prevention campaigns achieved a reduction of 1.46 and 2.5 kg/household a week respectively (Cox et al., 2010). The campaigns achieving these reductions have been criticised for only attracting especially interested inhabitants, still they show what is possible. Research is also done on the effect of changing to healthier or more sustainable diets (Alfredsson, 2004; Carlsson-Kanyama et al., 2003; Tukker et al., 2011).

Like most cities in a highly developed country today Trondheim has a well-developed urban solid waste management system, with a high efficiency regarding recovery and recycling of energy and materials. As a result, energy and materials from waste can substitute large quantities of energy and materials from other sources, and thereby avoid emissions and environmental impacts in a lifecycle and systems perspective. Other studies have reported that this substitution effect will actually result in net beneficial impacts (Christensen et al., 2009; Gentil et al., 2009). This is also the fact in Trondheim, where large amounts of paper, plastic, metal and glass waste are already source-separated for recycling, and where energy from incineration of residual waste is used for district heating (Brattebo and Reenaas, 2012; Slagstad and Brattebø, 2012).

Methodology

To estimate potential economy-wide environmental impact benefits of household waste prevention a methodology has been developed that includes activities both upstream and downstream to the households, see Figure 1. The upstream part of the system is analysed by use of an environmentally extended input-output (IO) framework, and the downstream part of the system is analysed by use of a solid waste LCA-framework.

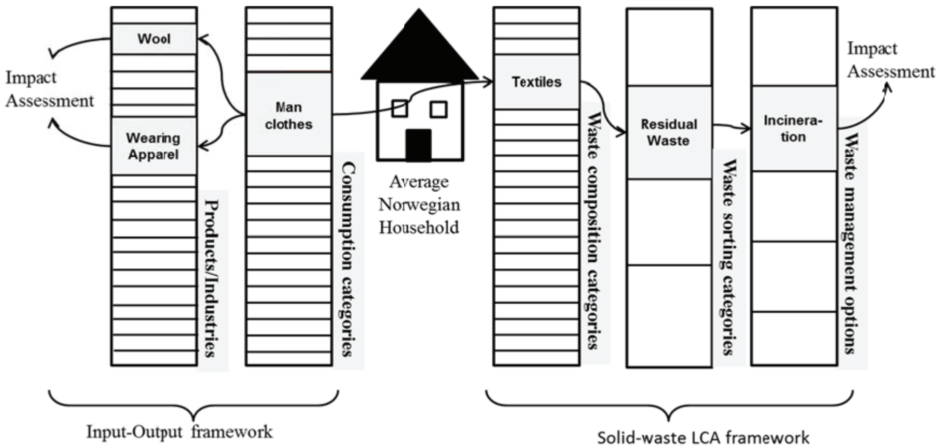

Figure 1. Economy-wide IO-LCA framework model for environmental impact assessment of household waste prevention.

The model

Environmentally extended input-output databases connect trades in an economy with the environmental impact for each trade. For the model in this article a multi-regional environmentally extended input-output database, EXIOBASE, built by the project EXIOPOL (2012) is used. From EXIOBASE the environmental impact associated with consumption of different product-groups (industries) for each country is extracted (import is included). To find the level of consumption from households, a consumer expenditure survey can be used. Since the consumption categories in the consumer expenditure survey are not necessarily the same as the product-groups in EXIOBASE, a connection between each of the data sets has to be made. By combining these two datasets the impact per unit of money spent in the household can be estimated. To avoid double counting the waste management expenses are removed from the household expenditures. In the Consumer Expenditure Survey, the “refuse collection” category is set to 0 Norwegian kroner.

The LCA-tool EASEWASTE (Kirkeby et al., 2006) is used as the solid-waste LCA framework. This is an LCA-tool designated to examination of waste management systems, where waste can be followed from collection to final destination, whether this is landfilling, incineration with heat recovery, or recycling with substitution of alternative processes. When performing LCA a functional unit has to be defined, and for waste management systems this is usually treatment of a given type of generated solid waste based on weight. Waste prevention challenges how to define the functional unit. To be consistent in a system-wide perspective, the quantity of prevented waste has to be considered as a virtual waste flow with no environmental burden in the waste system. This is explained in more detail in Gentil et al. (2011).

The total effect of household waste prevention is determined by combining the upstream input-output model with the downstream waste management model. However, one waste fraction (like paper, plastics or organic waste) can pull on numerous upstream product consumption categories, and some of these categories contribute to waste in several waste fractions. In addition, many of the product consumption categories are not contributing to household waste at all. Hence, only the waste fractions that are subject to actual household waste prevention must be linked with their relevant product categories upstream. At this stage linearity in the connection between the two sub-models for most waste fractions is assumed. This means that if the amount of a given waste fraction is reduced with 50 % due to waste prevention, the consumer expenditure of the related product categories will also be reduced by 50 %. This is believed to be a valid assumption since the turnover is relatively fast, and purchased goods will become waste within the same year. One exception is the food and drink category where additional factors have to be included. First, most of the food and drink purchased never becomes solid waste, as it is in our body transformed to energy, urine and faeces in wastewater. Second, while the food waste fraction only includes food waste, the upstream consumption categories of food and drinks also contain packaging. The share of packaging in the food consumption category therefore has to be known. The share of bought food becoming food waste differs between countries, the same holds for the share of packaging. These numbers therefore have to be adjusted to the country in question, if such numbers are available.

The combined IO-LCA framework can be modelled with and without rebound effects. Households can spend the saved money on other product categories, such as holidays and cultural activities, or the savings can be spread on a large number of other product categories. Three methods of accounting for the rebound effects are examined in this paper; i) to redistribute the money after a simple rebound principle, ii) to redistribute the money after a marginal rebound principle, and iii) to re-spend money on special chosen categories, as for second-hand stores, culture activities, holidays or restaurants.

The simple rebound principle distributes the money saved by reduced consumption in one category on all the other categories, proportionally to the relative expenditure size of the other categories. Extra money will thereby be used the same way as the money already available in the household. However, according to Alfredsson (2004), Swedish households would tend to spend extra available money in a different way than the money initially available in the household. To estimate how additional money will be used, the “marginal propensity to spend” principle can be applied. This principle uses consumer surveys in different income levels to estimate how a change in available income will change the consumption pattern of the household. The principle is explained in Alfredsson (2004), and according to her study Swedish households in given income levels would use extra available money on traveling and recreation, followed by food, and then clothes, housing, services, furniture and health.

Case study for applying this model

In relation to the development of a carbon neutral urban settlement in Trondheim city, the model described above was used to calculate the potential effect of household waste prevention on the economy-wide emissions of greenhouse gases. The Norwegian consumer expenditure survey (SSB, 2010) used in combination with the EXIOBASE, for upstream analysis, has 182 product consumption categories, while the EASEWASTE model, for downstream analysis, has 48 waste categories. There were no local consumer expenditure surveys available for Trondheim with the same aggregation level as the national survey; hence it was decided to use the national expenditure survey.

As this is a first attempt to develop a combined IO-LCA framework for examining waste prevention, it was decided to concentrate on the waste fractions that are fairly directly related to given consumption product categories. A set of scenarios was developed in order to analyse and compare the potential environmental impacts of assumed waste prevention outcomes, for different waste fractions and different rebound effect principles. The first scenario is the reference scenario, where there is no waste prevention and therefore no rebound effect. For all the other scenarios, we applied either the simple rebound principle, the marginal rebound principle, or the holiday rebound principle, as well as some special rebound effects related to each prevented waste fraction (see Table 1).

Table 1. Scenarios and rebound calculation principles.

No	Scenario		Chosen rebound calculation principle
1	Reference scenario	A	No rebound
2	50 % less food waste	A	No rebound
		B	Simple rebound
		C	Marginal rebound
		D	Holiday rebound
		E	Restaurant rebound
3	50 % less textile waste	A	No rebound
		B	Simple rebound
		C	Marginal rebound
		D	Holiday rebound
		E	Second hand + marginal rebound
4	50 % less paper waste	A	No rebound
		B	Simple rebound
		C	Marginal rebound
		D	Holiday rebound
		E	Cultural rebound

To calculate the factors for the marginal propensity to spend, the consumption profile for a household with average Norwegian income was compared with the consumption profile at the next income level. The difference between these two levels is then the estimate on how a consumer will distribute extra income, which is then the basis for how the given rebound effect gives cause to additional GHG emissions.

The waste composition used in the calculations is based on recent data for household waste in Trondheim: paper 24.3 %, food 24.8 %, textiles 3.9 %, plastic 12.1 %, glass 5.3 %, metals 3.9 % and others 24.7 %. Of the household waste fractions studied 70 % were incinerated with heat recovery, while the remaining 30 % were separated for materials recycling.

The 5 million inhabitants of Norway deliver about 420 000 tonnes food waste per year to the waste management system, and this represents a large share of the total household waste generation (NOK and LOOP, 2010). Reducing food waste is therefore one of the main waste policy priorities in Norway, and a research project examining how food losses in the value chain can be reduced started in 2010. As discussed earlier we cannot assume that a 50 % reduction in food waste leads to a 50 % reduction in consumption of food and drinks. According to a report prepared by WRAP in England 22 % of purchased food and drinks becomes waste in the UK (WRAP, 2009). The same relation is used for Trondheim, due to lack of such data locally. This means that by reducing the amount of food waste with 50 %, purchased food and drinks will be reduced with 11 %. We have not included the environmental impact of reduced drink waste to the sewer system. This assumption is based on the fact that the total impact from sewage treatment is low in Trondheim. A reduction in food consumption will, however, also reduce the amount of packaging in the household. Hence, a reduction in packaging waste has to be included when estimating the waste composition. In the food waste category a rebound effect is added with re-spending the money saved on reduced food and drink purchase on restaurants. Of the total consumer expenditure 11.8 % goes to food and drinks.

The second fraction where waste prevention is applied is for textiles, in this case including clothing and footwear. Of the total consumer expenditure 5.3 % goes to clothes and shoes. The amount of textile waste is reduced by 50 %; this will reduce the expenditure of textiles with 50 % as well. Most of the textile waste is incinerated, and less than 10 % are collected for reuse, carried out by two large organisations who export a large share of the collected textiles for sale in other countries. When it is assumed that some of the money saved on reduced consumption of new textiles is used in second hand stores, it is at the same time assumed that there are enough available second hand clothes and shoes in the stores. It is also assumed that the price of second hand textiles is half the price of new textiles, while the rest of the money is spread to the other consumption categories after the marginal propensity to spend principle. In this scenario the service level is kept constant by assuming that the same amount of clothes is bought from second hand stores.

The last fraction assessed is the paper waste fraction. Of the total consumer expenditure 1.5 % goes to paper consumption. Paper is, however, together with food waste the largest share of the household waste. Newspaper, magazines and advertisement are the most important fractions in weight. Paper waste has today high recycling rates, advantageous substitution possibilities, and advantageous calorific values when used for incineration with energy recovery. Still waste prevention is the objective. It is assumed a 50 % reduction in paper waste evenly distributed between the paper waste categories, and also a 50 % reduction in expenditure on paper.

The average waste amount generated by a household in Trondheim is estimated to 673 kg per year when source separated garden waste, EE-waste, hazardous waste and bulky waste are excluded (based on figures from 2010). The functional unit in our analysis is “collection and treatment of 673 kg household waste from one household in Trondheim during one year”. For all the scenarios an equal distribution between categories involved in the same prevention activity is assumed. This means that for food waste, both the vegetable food waste and the animal food waste is reduced with 50 %. The same holds for the product categories on the consumption side. For processes within Norway the Norwegian average electricity mix is used both in the waste management system and on the consumption side. Impacts are calculated based on global warming potentials in a 100-years perspective given by IPCC (2007) for both the input-output model and for the waste management system.

It has to be underlined that the assumed percentages of household waste generation being prevented in our study are chosen in order to illustrate the possibilities with the model and to demonstrate the relative importance of different elements of the system (product categories and waste fractions; upstream and downstream) with respect to contributions to system-wide GHG emission reductions. A closer discussion on what are realistic prevention targets, and how different waste prevention policies may actually change the generation of given household waste fractions, is outside the scope of this article.

Results

Let us start by examining what is the potential influence of waste prevention on GHG emissions from the waste management system, which is calculated by the “Solid waste LCA framework” part of the combined IO-LCA framework, see Figure 1.

The reference scenario was found to have a total impact of 4 kg CO_2-eq per household, see Figure 2. There are two main reasons for the low total impact from the waste management system. First, there are no emission-intensive wastes going to landfills in Trondheim today, since landfilling of organic waste was banned in Norway in 2009. Second, substitution of materials and energy is included in the LCA, and this can in many cases actually result in beneficial environmental impact from the waste management system (Christensen et al., 2009; Gentil et al., 2009; Slagstad and Brattebø, 2012). The household waste that is today incinerated is used for heat recovery and district heating, thereby substituting other energy sources for space heating in urban buildings (Brattebo and Reenaas, 2012). In Trondheim it is estimated that 70 % of this substituted energy is electricity, which is a typical situation for

Norway, but very untypical for most other countries. The Norwegian electricity mix is mainly based on hydropower and therefore fairly clean. The use of a Norwegian electricity mix is the reason for the detrimental impact in this system, in contradiction to other assessments where more emission intensive energy is replaced.

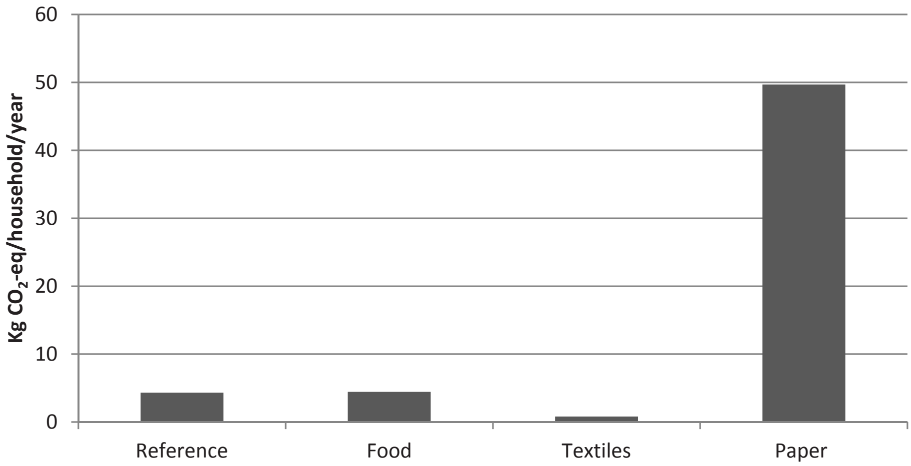

Figure 2. Total GHG emissions from the waste management system, for each waste prevention scenario.

When waste prevention with 50 % reduction in food, textiles and paper waste is analysed, the food and textile scenarios have very similar impact as the reference scenario, hence, food and textile waste prevention does not significantly influence GHG emissions from the downstream waste management system. Food waste is a relatively large share of the waste in Norwegian households; still reduced amounts of food waste give little change in impact from the waste management system. Food consists mostly of biogenic carbon and as long as the food is not landfilled, the environmental impact from waste treatment is very small. In addition above 70 % of the energy replaced by the heat produced in the incinerator is electricity based on hydropower, and a change in total heat recovered from the incinerator will change the total impact marginally. Food waste has also limited heating value compared to other fractions. A reduction in textiles waste will give less total impact due to the reduction in CO_2-emissions from the incinerator. Paper waste prevention will give an increase in total impact. This is mainly due to the reduced amounts of newspaper waste used for replacing virgin newspaper production. According to these results there are no benefits in the waste management system of applying waste prevention measures for food and paper, and only a very small benefit when preventing textiles.

Let us then examine what is the potential influence of waste prevention on GHG emissions from the total economy-wide system, using both the “Input-Output framework” part and the “Solid-waste LCA framework” part of the combined IO-LCA framework. While the total GHG emissions from the downstream waste management system in the reference scenario were 4 kg CO_2-eq per household, the total emissions from upstream household consumption are estimated to 22.8 kg CO_2-eq per year. Hence, the waste management system is of little importance with respect to the total impact generated in a household, given a situation with technologies, consumption and waste generation levels as in the case of Trondheim.

When the effect of waste prevention on the consumption side is included, the difference between the scenarios becomes much larger than what was reported in Figure 2, now see Figure 3. The no-rebound scenarios are the most beneficial ones for each waste fraction prevented. By reducing textile waste and textile consumption with 50 %, and saving the money (i.e. no-rebound effect is occuring), the total impact from households can be reduced with almost 950 kg CO_2-eq per household, which is roughly 4 % of the total GHG emissions from an average household. Similarly, 50 % reduction of food waste can reduce the total emissions with around 500 kg CO_2-eq per household. When the total change in emissions by reducing food consumption and textile consumption is compared, it must be remembered that while a 50 % reduction in textile waste leads to a 50 % reduction in consumption, a 50 % reduction of food waste only leads to an 11 % reduction in consumption, according to the model assumptions. For paper reduction the increased impact in the waste management system is followed by a relative low decrease in impact from changed consumption compared to the other fractions. In total 50 % reduction in paper waste will result in reduced emissions of approximately 50 kg CO_2-eq per household per year.

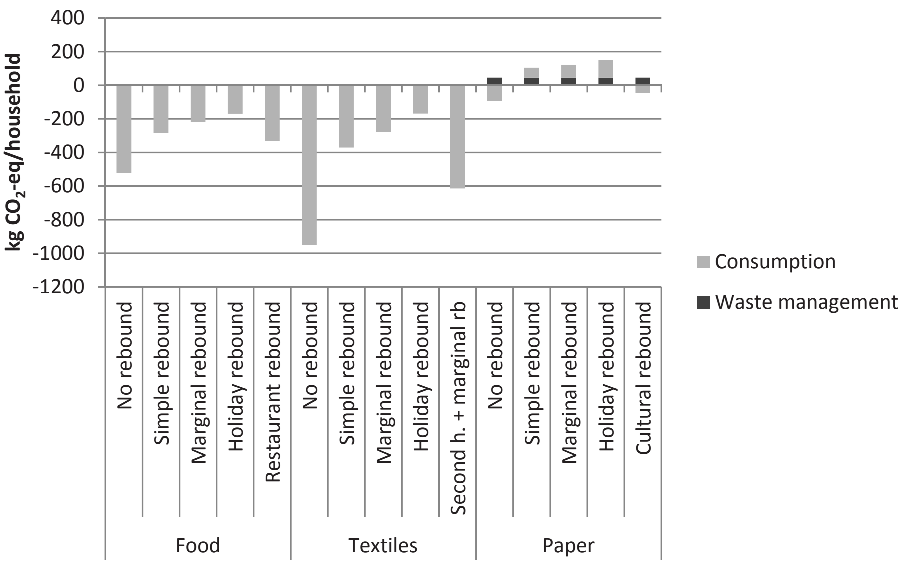

Figure 3. Combined IO-LCA influence of waste prevention on total GHG emission. All bars give the change in emissions relative to the reference scenario. Negative bars indicate reduced emissions compared to the reference scenario, while positive bars indicate increased emissions.

From this it can be seen that the rebound effect of waste prevention has a potential large impact on the economy-wide GHG emissions. If it is assumed that all the money saved on reduced consumption will be re-spent, it is important where the money is spent, due to the different emission intensity (kg CO_2-eq per Euro) of each product category. Two scenarios were analysed where the money saved were redistributed among all product categories, one scenario with all the money saved spent on holiday, and in addition some special rebound effects for each scenario. Re-spending the money on holiday is the worst-case scenario for all the prevented fractions included in our analysis. If money saved from food waste prevention is spent on holiday, the emission benefits (relative to the reference scenario) will decrease from 500 kg CO_2-eq per household to below 200 kg CO_2-eq per household. For textile waste prevention the benefit will decrease from 950 kg CO_2-eq to just above 150 kg CO_2-eq per household, and for paper waste prevention there is no benefit at all and the total GHG emissions will increase. Of the rebound effects tested in this article, the best way of re-spending money-savings from waste prevention is to use more on restaurants, second-hand stores, repair and cultural activities.

If all the above targets on waste prevention were achieved, a 27.5 % reduction in total waste from a household would result in the benefit of 6.7 % reduction in total economy-wide GHG emissions, in a no-rebound scenario. If the rebound effect with all the saved money were re-spent, a holiday scenario would decrease the waste prevention emission benefit to 0.8 %, while money re-spent on cultural activities will give an emission benefit of 5.3 %. The marginal and the simple rebound effects are in the middle of these two other rebound effects.

Discussion

Our combined IO-LCA framework model is in its early development phase and there are of course uncertainties in the results, especially related to how to link the different data sources (such as the environmentally extended input-output data, the consumer expenditure surveys and the different waste fractions). For the Trondheim case, the EXIOBASE with 129 categories had to be combined with the consumer expenditure survey with 182 categories. Moreover, the waste management system had only 48 waste fractions, and the waste composition was estimated based on a waste composition analysis comprising 42 waste categories, information from the waste company and amounts of sorted fractions from Statistics Norway. At the present stage of IO-LCA modelling, the results of the assessment for Trondheim will only give an indication on the levels of importance for the different scenarios including the rebound effect.

Regarding food waste, the objective in this article has been to examine the potential effect of reducing the amounts of food waste, and not necessarily how waste prevention will affect the change in diet. The prevented food waste is therefore evenly distributed on all food consumption categories and on all food waste categories. As mentioned earlier in the article, over 50 % of the food thrown away has at some time been eatable, and therefore, strictly speaking, a 50 % reduction in food waste could be achieved by eating the food already in the household. However, it is found more likely that a reduction in consumption of food has to be included. In England two different food prevention campaigns achieved a reduction of 1.46 and 2.5 kg per household a week, respectively (Cox et al., 2010). The campaigns achieving these reductions have been criticised for only attracting especially interested inhabitants, but still they show what is possible. In the case study applied in this article 3.2 kg food waste per household a week

In Norway the prices of clothes have fallen at the same time as the relative income has increased, and this has led to higher consumption of clothes and an increase in the amounts of textile waste (Laitala and Klepp, 2011). Clothing was in an EU report found to contribute with between 2 and 10 % of a consumer's environmental impact (European Commision, 2006). In this article consumption of clothes contributes with 8 % of the GHG emissions, which is in the upper range of the interval reported for the EU. There are different ways of reducing textile waste, with and without including the rebound effects. Buying less clothes, but with higher quality, is often pointed at as the most preferred solution. In this way the money saved on reducing the quantity of clothes are used to increase the quality of the clothes. The comparison of the environmental effect of producing low-quality textiles with high-quality textiles is, however excluded from this assessment. The use of second-hand stores is a good environmental choice according to the results. This is supported by findings of Woolridge et al. (2006) who found reuse of polyester garments and cotton clothing through second-hand stores to require less than 3 % of virgin textile production. Buying textiles at second hand stores will, however, not reduce the impact related to the use phase of textiles (e.g. washing), which, depending on the energy source, can be an important contributor to textiles life cycle impact.

According to our analysis, paper prevention has smaller environmental benefits than textiles and food. Gentil et al. (2011) also found prevention of food waste being more important than paper waste prevention, when using an LCA-extended model for calculating the environmental effect of waste prevention. An evaluation of which consumption categories should be targeted by households in Norway found clothes and food to be amongst the most important ones (Raadal et al., 2006). This is confirmed in the IO-LCA analysis. When the rebound effect is included, all the paper prevention scenarios become less beneficial than the reference scenario, except when the saved money is used on cultural activities.

Gentil et al. (2011) used the LCA-extended model to calculate the effect of preventing unsolicited mail. This waste fraction is difficult to include in our present model, because the households do not directly spend money on the mail, but pays for it indirectly by buying other goods. It is not possible to extract this fraction from the input-output table at the aggregation level used in this model. This is one example of a present shortcoming in the IO-LCA model used in this study. It will not work for all household waste fractions one could possibly prevent. For now, only some of the waste fractions can be fairly easily linked to the input-output product categories. Furthermore, the connection between savings in one product category and additional (rebound) expenditure in another is only assumed. The risk of double counting is another challenge when combining two different models for the estimation of GHG emissions, due to the difference in aggregation level in the two models. By excluding the consumer expenditure on household waste all the processes included in the LCA-framework are assumed excluded from the upstream total impact. This is, however, difficult to prove at this stage. At the level of detail in the present study double counting is avoided as far as possible, but it will be an important issue to address when the model is developed further.

The results in this study clearly demonstrate the importance of the rebound effect. The worst-case scenario, of all the scenarios tested in this article, is to re-spend the saved money on travelling. The importance of the rebound effect is also discussed by Alfredsson (2004) which found ‘green consumption’ to have limited effect. The same was found by Druckman et al. (2011). It is difficult to estimate this effect related to waste prevention when applying traditional LCA-based models only. Hence, it is believed that assessment of environmental benefits due to waste prevention strategies indeed has to use IO and LCA methods in a combined IO-LCA framework, for instance such as the one used in this study. However, since there are several assumptions and simplifications to be made, also in the presented study, significant further research are needed in this area. In fact, it is a striking observation that the bulk of literature on LCA and environmental impacts from waste management hardly examines waste prevention, per se, even though prevention is unanimously accepted as the top priority in the waste management hierarchy.

Conclusions

Waste prevention is a top priority in the present waste management hierarchy. However, little research is done on how waste prevention affects the economy-wide environmental impacts. This study developed a combined IO-LCA framework to estimate the potential GHG-emission benefits of waste prevention, and the likely implications of rebound effects from money savings due to shifts in household expenditure as a result of waste prevention.

If the aim is to reduce economy-wide GHG emissions, the upstream production phase of goods seems to be much more important than the downstream waste management phase, at least as long as we are in a country with high-quality waste management. We need economy-wide models to calculate the environmental effect of waste prevention, and these models should take the advantage of combining IO methods on the upstream side with LCA methods on the downstream side. The combined model then need to link product/consumption groups and waste fractions, with the use of available statistics. There are advantages and disadvantages with using IO analysis to assess the upstream impacts. The possibilities to include the rebound effect are seen as one of the most important advantages. One disadvantage is the aggregation level in the analysis, which can be challenging when looking at special product groups or waste fractions.

For the case of Trondheim we can, however, already at this stage conclude with relative large upstream benefits of preventing textile and food waste. The rebound effect is found to be very important, and it can to a large extend offset the GHG-emission benefits by waste prevention, especially if the extra money is used on travelling. Where saved money is actually re-spent is therefore important. The member states of the EU shall by December 2013 establish a national waste prevention programme (European Commision, 2008). We believe that a combined IO-LCA framework and GHG-emission model, like the one we developed in this study, can effectively contribute to estimate the potential consequences of waste prevention with respect to GHG emissions, including how the rebound effect of changed consumption expenditures can influence the emissions and benefits of waste prevention programmes.

Acknowledgement

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Paper 4

Life cycle assessment of the water and wastewater system in Trondheim, Norway – A case study

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Life cycle assessment of the water and wastewater system in Trondheim, Norway – A case study

Helene Slagstad and Helge Brattebø

Abstract

This study presents the results from a life cycle assessment (LCA) performed on the water and wastewater system in the city of Trondheim. The objective of the study was to examine the system-wide life cycle environmental impact potentials of operating the city's water and wastewater system, in order to clarify the relative importance of different environmental impact categories and how different elements of the water and wastewater system contribute to these impacts. As the results of this study were used in the planning of a new carbon -neutral urban settlement, the climate change impact was of special interest. Freshwater eutrophication due to the consumption of energy and chemicals was found to be the impact category with the largest contribution to the total environmental impact. In practice, urban water utilities would have to perform a trade-off between the consumption of energy and chemicals and the discharge of pollutants to receiving waters.

Keywords: carbon neutrality, life cycle assessment, urban water infrastructure

Introduction

The services provided by urban water and wastewater utilities are based on legislations on water supply and wastewater management, including standards for water quality and pollution discharge to the local receiving water bodies. Utilities commonly have a strong focus on water quality, treatment efficiencies and cost-effectiveness; however, during recent years increasing focus is given to wider sustainability criteria, including carbon dioxide emissions and life cycle environmental impacts.

Water and wastewater infrastructures play an important role in daily urban life. To perform this role the system consumes materials, chemicals and energy for the construction, operation and maintenance of treatment plants, pipeline networks, reservoirs and pumping stations, all of which are associated with environmental impacts. There are different methodologies available for estimating the environmental impact of these systems. When analysing the assessment of recycled water schemes, Chen et al. (2012) compared the use of Material Flow Analysis (MFA), Life Cycle Assessment (LCA) and Environmental Risk Assessment (ERA). They found that MFA is an effective initial screening method, that LCA is widely used in finding the optimal wastewater treatment technology and that ERA mainly evaluates site-specific chemical hazards. Stokes and Horvath (2011) demonstrated how estimating the environmental impact of water systems without including the life cycle impact of energy and materials can significantly underestimate the total environmental effect of the system. The advantage with LCA is that it not only takes direct emissions into account, but also includes impacts resulting from production and transportation of resources, construction and maintenance of buildings and infrastructure, end-of-life management, etc.

LCA has been used in the water and wastewater research for some time (Godskesen et al., 2011, Lundin et al., 2000, Ortiz et al., 2007, Remy and Jekel, 2008, Stokes and Horvath, 2010, Stokes and Horvath, 2011). Yet studies focusing on the entire water and wastewater system are relatively few in number. Among those studies, Lassaux et al. (2007) examined the water and wastewater system in the Walloon Region in Belgium. They found that the environmental impact of the water system was less than the environmental impact of the wastewater system, and that the most important environmental strains were derived from water discharge, wastewater treatment operations and, to a lesser extent, the sewer system. Venkatesh and Brattebø (2011) developed a 'metabolism model' for urban water systems, and studied the energy consumption, costs and environmental impact of urban water cycle services in Oslo. Their study demonstrated that the wastewater treatment plants have the highest environmental impact, most notably from acidification and eutrophication. After weighting, they found that global warming accounted for only 6% of the total impact score when considering the operation and maintenance phase of the system. Lundie et al. (2004) conducted a prospective LCA on the water and wastewater system of Sydney, Australia, as a basis to recommend measures for improving the system's environmental performance.

In Trondheim, Norway, a new 'carbon-neutral' urban settlement is planned, at Brøset. The average annual global -warming impact in Norway is 14.9 tonnes of CO_2-eq per person (Hertwich and Peters, 2009). In effort to achieve carbon-neutrality at Brøset, every part of the project, including the water and wastewater system must contribute to impact reduction. There has been limited knowledge of the impact of conventional water and wastewater systems in Norway, and before new alternative solutions at Brøset were suggested the conventional system in Trondheim had to be thoroughly examined. The objective of this study was to quantify the system-wide life cycle environmental impact potentials of operating the city's water and wastewater system, in order to clarify the relative importance of different environmental impact categories and how different elements of the water and wastewater system contribute to these impacts. Particular focus is given to the system-wide carbon dioxide emissions and contributions to climate change, since this is in general a high priority issue for water utilities, and for Trondheim in particular due to the planning of the new urban settlement with the ambition of carbon-neutral solutions. Possible improvements to the system will be discussed, but not assessed at this stage. We claim that more knowledge and better methods are needed for assessing the environmental impact of water and wastewater systems, and this case study helps to expand the knowledge about the environmental impacts of water and wastewater systems in urban areas.

Methodology

Life cycle assessment

In accordance with the literature, we decided that LCA is the best method for assessing system-wide environmental impact potentials of the current water and wastewater system in Trondheim. There are other tools available for assessing environmental impacts of different systems; however, due to its unique and comprehensive life -cycle perspective, LCA is found superior to other methods, such as Strategic Environmental Assessments, Cost-Benefit Analysis, Material Flow Analysis, Environmental Risk Assessment, or Ecological Footprints (Chen et al., 2012, Finnveden et al., 2009). Life cycle assessment is standardized (ISO, 2006a, ISO, 2006b), and commercial LCA-software programmes are mature and robust (Chen et al., 2012). LCA is effective for evaluating the environmental impacts of the systems under study, but the methodology also has constraints. LCAs can be data intensive and time consuming, and deciding what should be included or excluded in the assessment is therefore important. Hence, setting the system boundaries can be challenging; leaving out processes assumed to be of minor interest in the study at hand can result in the omission of significant impacts. Moreover, LCA employs generic characterisation factors for local or regional impacts, such as eutrophication. As will be discussed in this case study, so long as these characterisation factors are not regionalised, results must be interpreted in the light of local conditions. Work is undertaken to improve the accuracy of regional impact categories. In addition to these factors we have to deal with uncertainty in the parameters used in the assessment.

The functional unit of our study consist of a one-year provision of water, and collection, transportation and treatment of wastewater (including stormwater) for Trondheim, Norway. The system boundaries are given in Figure 1. The LCA -programme Simapro version 7.3.2 (Pré Consultants, 2011), with the Ecoinvent database was used for the assessment. Ecoinvent has life cycle inventory data on energy supply, resource extraction, material supply, chemicals, metals, agriculture, waste management services, and transport services. These data are combined with data for energy and material use and data for embodied energy calculations for buildings, pipelines, pumps, and water storage devices. Emissions from overflow, effluent and sludge, fertiliser substitution, and transportation were also accounted for. Impact categories dealing with toxicity, however, were excluded from the study due to lack of data on toxic elements in effluent, overflow, and sludge. For the impact assessment, the midpoint impact assessment method ReCiPe (midpoint (H) v1.06, July 2011), with normalisation values for Europe, was applied. ReCiPe 2008 builds on the Eco-indicator 99 and the CML Handbook on LCA, and is an impact assessment method harmonised with respect to modelling principles and choices concerning midpoint and endpoint impact assessments (Goedkoop et al., 2012). The processes included in the assessment are given in the Appendix.

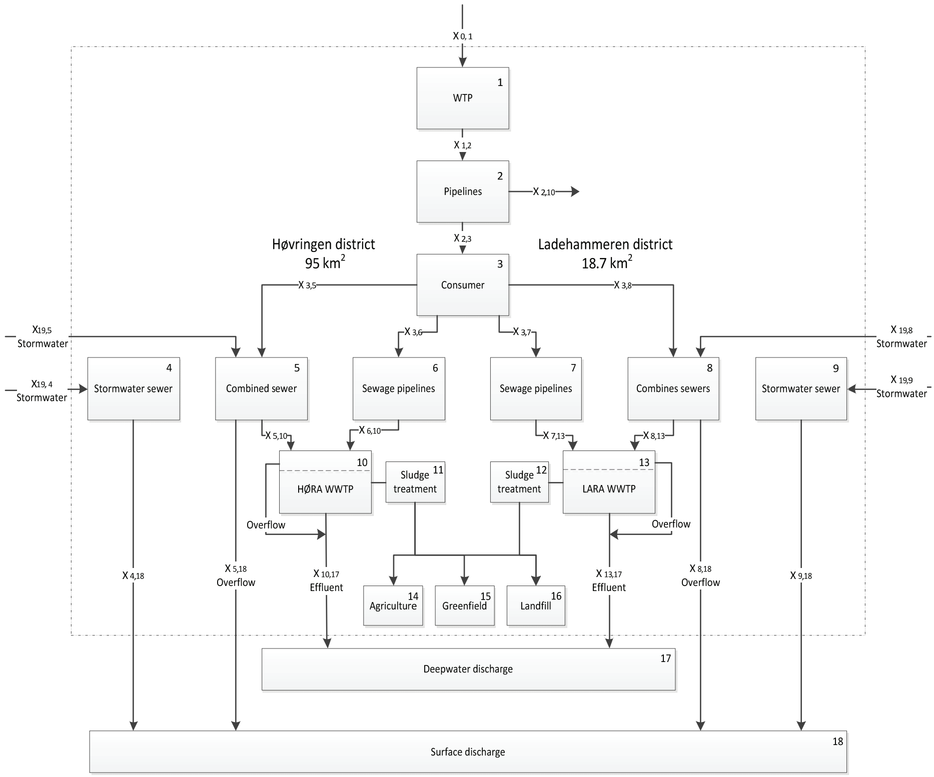

Figure 1. Water and wastewater flow diagram for Trondheim.

Case system description

Trondheim is the third largest city in Norway, with 171,000 inhabitants. The water is supplied from surface water collected from a large nearby lake called Jonsvatnet (Figure 2). Water is treated in a central water treatment plant at Vikelvdalen (VIVA), then distributed for consumption, followed by collection of stormwater and wastewater for treatment in one of the city's two wastewater treatment plants - one at Høvringen (HØRA) and one at Ladehammeren (LARA). After treatment the effluent is discharged into the fjord. Stormwater is either collected in a separate pipeline system before being directly discharged into the fjord, or is sent for treatment together with wastewater in a combined sewage pipeline network. The system is described in more detail below. The data collected for the system were from 2010.

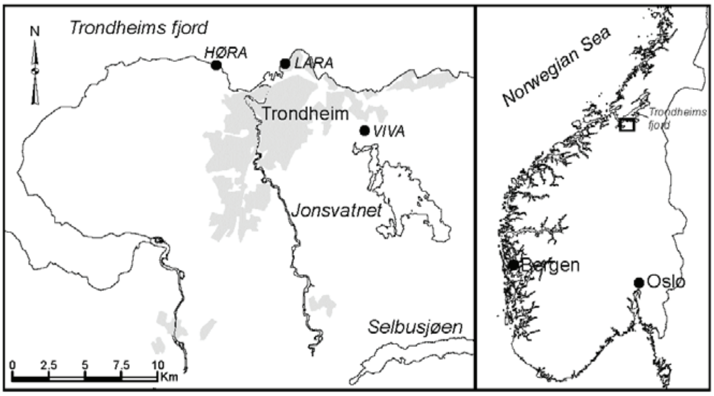

Figure 2. The case study area.

Potable water production and distribution

VIVA treated 22.3 million m^3 of water in 2010, a small fraction of which was also supplied to neighbouring municipalities. The facility used calcite, sodium hypochlorite (which is produced at the plant), carbon dioxide, and UV-filtration for treating the water to approved quality standards. This is a simple but efficient water treatment method suitable for producing good quality surface water. The municipality of Trondheim encompasses a surface area of 342 km^2, and 3.5 GWh was used for the 22 pumps in the water distribution network. Twelve water storage tanks were connected to the system, and all were accounted for in the calculations. A significant percentage of treated water was lost by leakage from the pipeline network, as some of the pipelines were nearly 150 years old and the level of maintenance has been low for many years. In fact, even some of the pipelines installed as late as the 1960s and 1970s were of poor quality - especially the ductile iron pipes which have not been coated for corrosion protection. Water leakages accounted for approximately 32% of the treated water that originated from the treatment plant. In other words, only 13.9 million m^3 of the 20.5 million m^3 of potable water produced for Trondheim was actually available to the consumers.

Pipelines

By calculating the masses of pipes, based on known lengths, diameters, and materials of construction, a detailed study was performed on the embodied energy in the 1900 km of pipelines in the public network (as distinguished from the private network). Assumptions about pipe thickness based on pipe diameter and material used were sourced from Venkatesh (2011). A diesel consumption of 29.35 MJ/m was assumed for installation - the same as that used in the Ecoinvent database for pipe installations. The lifetimes assumed were based on local knowledge, with 100 years for concrete, asbestos cement, and ductile iron pipes, 70 years for steel and copper pipes, and 120 years for PE, PP, PVC, synthetic fibre, and glass fibre pipes. For wastewater pipelines, concrete was by far the most dominant material by mass. For water pipelines, it was ductile iron, although concrete was by no means insignificant. Maintenance of the pipes was not included in the assessment.

Consumers
The use phase of the water was omitted from our analysis. This means that private pipes, other in-house installations, and energy for water heating were excluded from the calculations. It was assumed that the volume of potable water entering the system was equal to the volume of wastewater discharged by private, public, and industrial consumers. Therefore, it was assumed that consumers expelled 13.9 million m^3 of wastewater annually.

Wastewater
Of the wastewater pipeline network in the city, 60% of the pipes were part of a separate system (sewage and stormwater flowing separately), although 10% of these pipes connected to combined sewers downstream. This, in effect, means that 50% of the sewage entered the wastewater treatment plants (WWTPs) through combined sewers and 50% through dedicated sewage-carriers. The Høvringen wastewater treatment plant (HØRA), with a catchment area of 95 km^2, treated 20.6 million m3 of wastewater in 2010, while the Ladehammeren wastewater treatment plant (LARA), with a catchment area of 18.7 km^2, treated 11.1 million m^3 of wastewater in 2010. About 50% of the wastewater was from industrial consumers. The wastewater at LARA was more concentrated when compared to HØRA, due to a smaller amount of stormwater entering LARA at an almost equal hydraulic load measured in Person Equivalents (PE). To find the amount of stormwater that entered the two plants, some estimates had to be made because of the lack of available data on flows between the input to consumers and the input to the WWTPs. The estimates were based on the volume of water going to the consumers, the volume of wastewater entering the WWTPs, the PE connected to each plant, and the approximate distribution of 50% for the separate system and 50% for the combined sewers. The complete system is depicted in Figure 1. The notations refer to the different flows in the system.

HØRA is a mechanical treatment plant using polymers for improved sedimentation and dewatering. It had a BOD5 reduction rate of 49.2%, and a tot-P reduction rate of 25%. $X_{5,10}$ (inflows to the treatment plant from combined sewer pipelines) was estimated to be 17.8 million m^3, and $X_{6,10}$ (inflows to the treatment plant from separate sewage pipelines) was estimated to be 2.8 million m^3. The plant consumed 2.3 GWh of electricity and 0.17 million litres of oil per year, and it produced 0.57 million Nm^3 of biogas, which was used internally for heating the sludge. Complete combustion of the biogas with no emissions of hydrogen sulphide, carbon monoxide, and ammonia was assumed.

LARA, on the other hand, is a chemical treatment plant, with a reduction rate of 45.3% of BOD_5 and 80.1% of tot-P. In this plant, 8.3 million m^3 of wastewater entered through combined sewers ($X_{8,13}$), and 2.8 million m^3 consisted of untreated sewage entering from the separate system ($X_{7,13}$). Iron chloride, together with polyamine and polymer, was used for sedimentation, and polymer was also used for dewatering. The plant used 2.2 GWh of electricity and produced 0.8 million Nm^3 of biogas. Around 60% of the biogas was used internally for heating, while 40% was used for hot water production, which was delivered to the district heating system in city. Thus, the hot water used in the district-heating system avoided the use of an annual average energy mix consisting of 72.4% electricity, 18.5% fuel oil, 5.2% wood, and 3.9% natural gas. As we can see, electricity (predominantly from hydropower) is the main heating source in Norway. Norway is, however, part of a Nordic electricity market, and we therefore choose to use a Nordic electricity mix in the calculations. This choice was tested in the sensitivity analyses.

There are three main outflows from the WWTPs – effluent, overflow, and sludge. The overflow and the effluent from the WWTPs enter the Trondheim fjord, which is connected to the Norwegian Sea. The Norwegian Sea, as well as the Trondheim fjord, is considered to be robust in terms of eutrophication; still large WWTPs are expected, by regulation, to have secondary treatment if special permissions have not been conferred, as in the case of Trondheim. The phosphorous concentration in the inflow and outflow of the plant was known, and therefore it was easy to calculate the phosphorous content of effluent, overflow, and sludge. Nitrogen content, on the other hand, was not measured, and some assumptions had to be made. According to standards used by Statistics Norway, the ratio between phosphorous and nitrogen is 1.6:12 (SSB, 2010). The nitrogen amount in the wastewater was therefore estimated based on the known phosphorous content. Treatment efficiencies for mechanical and chemicals plants were taken from Venkatesh and Brattebø (2009). While the nitrogen entering the sea has eutrophication potential, nitrous oxide (N_2O) is a greenhouse gas contributing to global warming with a factor of 298 times greater than CO_2 (IPCC, 2007). The 2006 IPCC Guidelines for National Greenhouse Gas Inventories estimates the emission factor of N_2O to be 0.5% of the nitrogen content of the effluent (IPCC, 2006). The uncertainty is great, however, with a range from 0.05% to 25%. For the time being these are the best estimates available. Still it must be borne in mind, that the emissions estimated were based on the assumption on the nitrogen content in influent, treatment efficiencies, and the calculation methods for N_2O-emissions. Therefore, there is strong uncertainty associated with N_2O-emissions.

Nitrogen and phosphorous in the sludge have value as a fertilizer, and can substitute the use of mineral fertilizers in agriculture. There are, however, several quality criteria for sludge, which separate it into three categories: sludge for agriculture, sludge for greening, and sludge for deposition. It is uncommon to incinerate sludge from WWTPs in Norway, and this is not done in Trondheim either. Of the 4155 tons of sludge from HØRA, 83% was used in agriculture, 16% was used for greening, and 1% was deposited. At LARA, of the 4309 tons of sludge produced, 69% was used in agriculture, 30% for greening, and 1% was deposited. Plant availability of nitrogen and phosphorous were assumed to be 50% and 70% respectively (Remy, 2010), and the fertilizer substituted was assumed to be Super Phosphate and Urea. Therefore, 12,700 kg Super Phosphate and 34,900 kg Urea were substituted.

Two additional flows had to be considered: the flow from the separate stormwater pipelines ($X_{4,18}$ and $X_{9,18}$), and the overflow from the combined sewer pipelines ($X_{5,18}$ and $X_{8,18}$). For the stormwater system, the pipelines and the pumping energy were taken into consideration; the environmental impacts associated with the flow of the stormwater into rivers/fjord were not accorded much importance in this analysis. The overflows, on the other hand, occur when the combined sewer system is overloaded because of heavy rain or failures in the system: untreated sewage then enters rivers. This is a problem since the rivers enter the fjord in shallow waters near popular beaches, which tend to get contaminated with bacteria after heavy rainfall. The municipality has decided on a policy to reduce the overflow into the fjord by rehabilitating the existing pipe network, and it is assumed that about 6% of the wastewater in combined sewers is presently discharged as overflows annually. Data on the exact concentration of these overflows was not available; an even distribution of untreated sewage and stormwater over the year was therefore assumed. In the HØRA catchment area an overflow of 1.1 million m^3 ($X_{5,18}$) was assumed, while the LARA catchment area gave an overflow of 0.5 million m^3 ($X_{8,18}$). The eutrophication potential takes these overflows into account.

A small wastewater treatment plant treating less than 1% of the water in town was omitted from the calculations due to lack of reliable data. Emissions associated with the spreading of sludge and mineral fertilizer were also excluded, on the assumption that these processes have negligible environmental impacts.

Results and discussion

Global warming

As the results of this study were used in the planning of a new carbon -neutral urban settlement, the climate change impact was of special interest. The combination of water treatment, piping and pumping of potable water, piping and pumping of wastewater, and wastewater treatment had an annual total impact of 8.2 million CO_2-eq, or 48 kg CO_2-eq per capita. The WWTPs had the largest impact (54%), with multiple sources like energy and chemical use (iron chloride), N_2O-emissions, and use of materials (Figure 3). Energy use contributed to 37% of the total impact on climate change for the entire system and was the most important contributor in the water treatment plant, water and wastewater pumps, and the HØRA wastewater treatment plant.

At LARA, emissions due to production of chemicals were more important, as this plant uses chemicals for treating the wastewater and in addition delivers energy to the district heating system in Trondheim. Therefore, some of the energy retrieved from biogas production at LARA offsets the impact from other energy sources. When upstream and downstream impacts were compared, the upstream contribution to global warming was found to be 17 kg CO_2-eq per capita annually (35%), while downstream impact was 31 kg CO_2-eq (65%). Water and wastewater pumps and storage contributed to 16% of the total impact, while the pipelines contributed 12%. The infrastructure of the water and wastewater system therefore contributed significantly to the total impact from the system, which is similar to the findings of Lassaux et al. (2007).

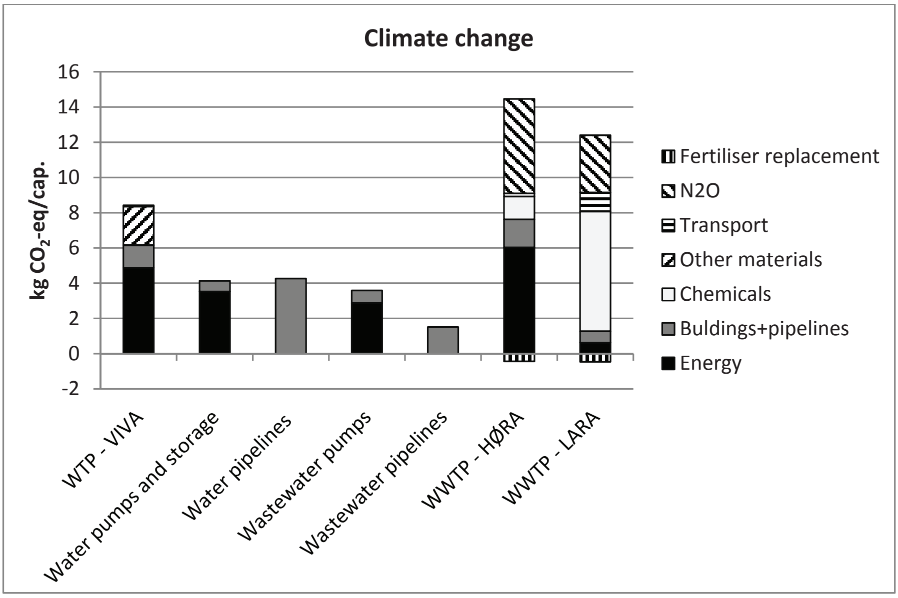

Figure 3. Climate change from the water and wastewater system in Trondheim (171,000 persons).

The normalisation value in ReCiPe for climate change is 11.2 tonnes of CO_2-eq annually per person in Europe, while according to Hertwich and Peters (2009) Norwegians have an annual carbon footprint of 14.9 tonnes of CO_2-eq per person. The contribution from the water and wastewater system to the annual total impact per person was in both cases less than 1%. In the planning of a new carbon-neutral settlement the impact from the water and wastewater system, if connected to the conventional system, is of minor importance. Improvements in impact may be possible, however, by reducing the impact of the entire system or by introducing alternative local solutions with reduced environmental impacts. When examining the water and wastewater system in Sydney, Lundie et al. (2004) found that increased water demand management, energy efficiency, energy generation, and additional energy recovery from bio-solids improved all environmental. Guest et al. (2009) and Larsen et al. (2009), on the other hand, call for a paradigm shift in wastewater handling. They propose improving resource recovery by moving away from conventional end-of-pipe solutions to source-separation technologies. Remy and Jekel (2008), by contrast, found that source - separation of wastewater does not necessarily result in a system with less environmental impact. Moreover, with the help of MFA and LCA, Jeppsson and Hellström (2002) also found it difficult to prioritise between high-tech, end-of-pipe solutions and source-separation strategies. Obviously, there are no easy solutions for reducing the impact on climate change from these systems, and according to our study, concerns other than water-related greenhouse gas emissions are more important to address in the planning of a carbon-neutral settlement.

Other environmental impacts

In a water and wastewater system there is a variety of environmental impact categories that should be considered. However, from our results it can be seen that ozone depletion, photochemical oxidant formation, particulate matter formation, terrestrial acidification, mineral resource depletion, and fossil resource depletion all had less than a 1% impact compared to the European average per-capita impact. The WWTPs contributed to more than 45% of the impact in each category; nevertheless water treatment, pumping and pipelines construction were all contributing to the total impact from the system in each category (Figure 4).

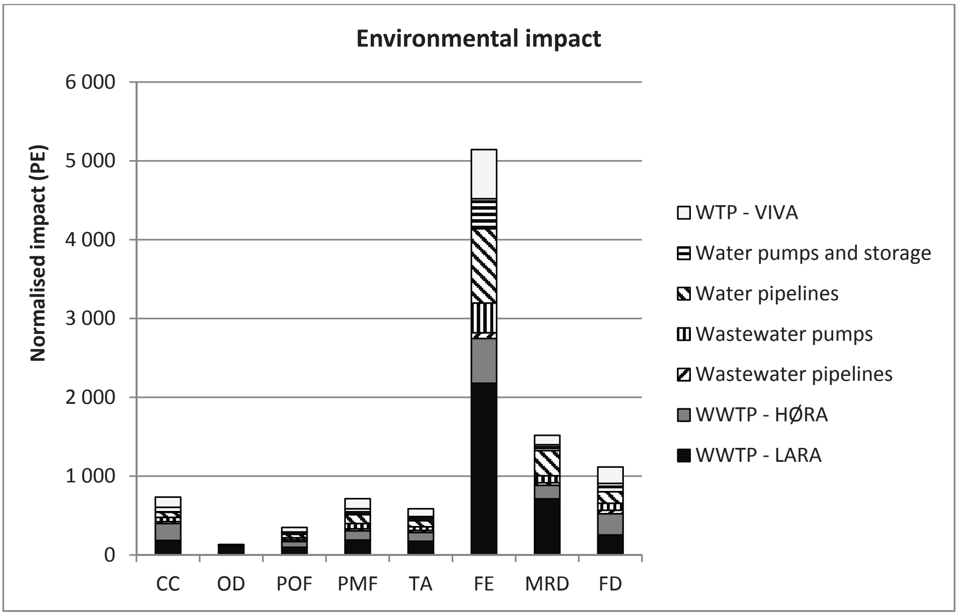

Figure 4. Normalized environmental impact, relates to 171,000 inhabitants in Trondheim. Climate change (CC), ozone depletion (OD), photochemical oxidant formation (POF), particulate matter formation (PMF), terrestrial acidification (TA), freshwater eutrophication (FE), mineral resource depletion (MRD) and fossil resource depletion (FD).

The LCA identified freshwater eutrophication as by far the most important impact category. In the case of Trondheim, the wastewater effluent and overflow was discharged more or less directly to a seawater fjord (see Figure 2). Therefore, the LCA freshwater eutrophication impact values are not a result of direct emissions from wastewater, but mainly a consequence of indirect emissions from coal mining in the production of electricity, due to the coal share of the Nordic electricity mix. Hence, local freshwater eutrophication does not have the potential to be a local urban pollution problem in Trondheim itself, but elsewhere in the value chain of electricity production.

Marine eutrophication results are not included in Figure 4. However, our calculations showed that this was the category with the theoretically largest environmental impact potential, since only a small part of the nitrogen content in wastewater is removed in the wastewater treatment plants in Trondheim. Eutrophication problems may occur in a marine fjord if this has little access to fresh, oxygen-rich water. In the case of the Trondheim fjord, however, this is not a concern, as it is 130 km long, several hundred meters deep, and with excellent exchange of fresh seawater with the outside Norwegian Sea. Thorough investigations of the fjord have demonstrated that it has excellent environmental conditions, except some local environmental hazard issues originating from other sources (Oceanor, 2003). The conclusion is therefore that the effluent from WWTPs in Trondheim can be safely emitted as it is, without a need for investing in improved high-grade treatment. As explained earlier the present LCA methodology does not take local conditions into account, but instead only gives an estimate on the potential for eutrophication, and without local or regional parameterisation. This problem is an acknowledged one, and work is currently done in the international LCA community to develop parameterised LCA -tools that use regional/local characterisation factors. Until such LCA- tools are available, interpretation of LCA results in accordance with local conditions is very important when analysing impacts from urban water and wastewater systems.

Figure 5 shows how the normalised total environmental impact is distributed between the different parts of the water and wastewater system. This is actually the same data as given in Figure 4, but presented in a different way for a clearer illustration of the relative importance of each part of the system. Figure 6 shows the importance of different resource inputs.

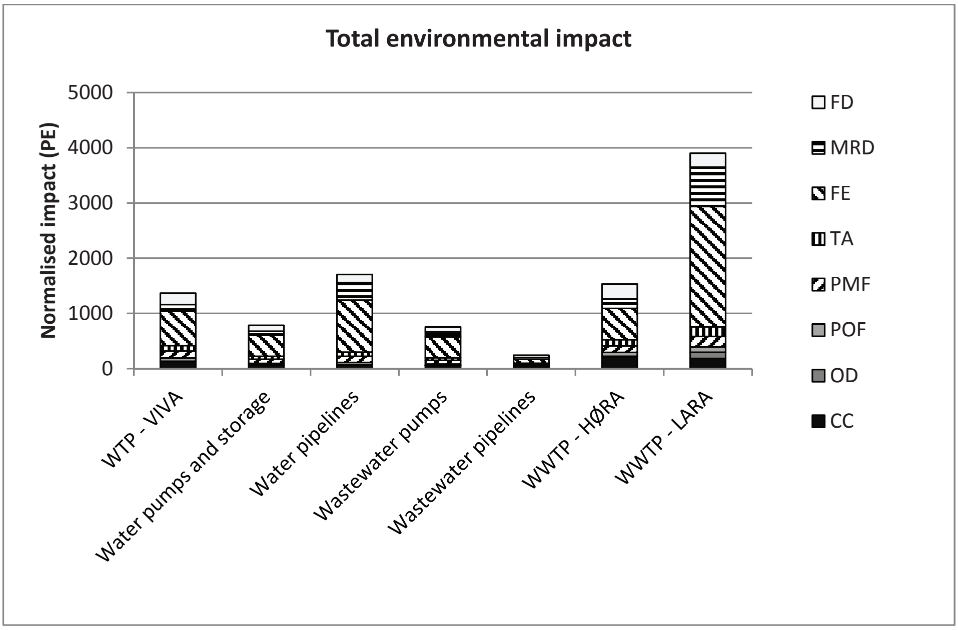

Figure 5. Normalised environmental impact – contribution of different parts of the system.

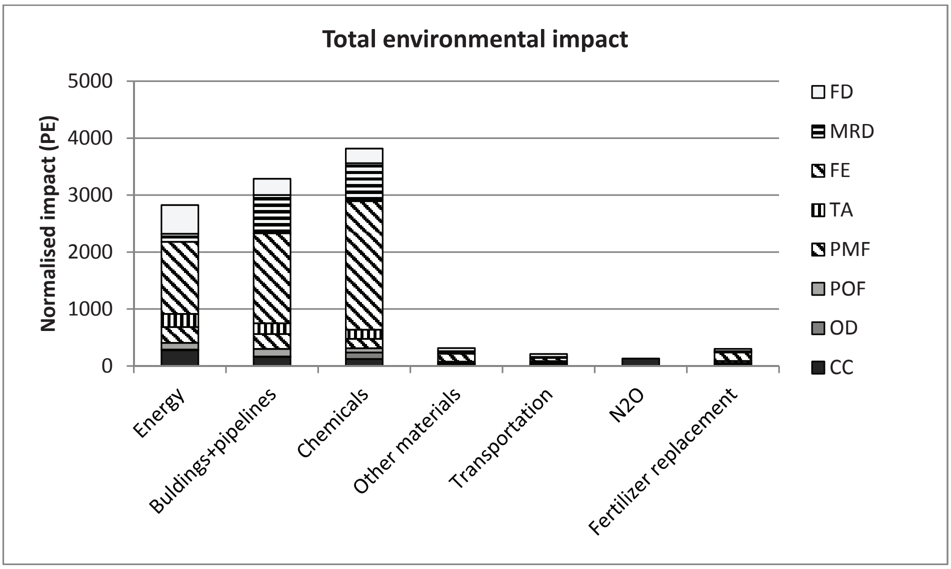

Figure 6. Normalised environmental impact – contribution of different resource inputs.

The results presented in Figure 4, 5 and 6, as a whole, provide an excellent demonstration of the usefulness of LCA, when aiming for system-wide environmental improvements within an urban water and wastewater system.

First, the LCA method clearly demonstrates that several environmental impact categories should be paid attention to in the urban water sector. A common priority of water utilities is reduced pollution of local receiving waters by use of advanced wastewater treatment plants. This case study for Trondheim shows that such a priority is sometimes not a good strategy, when dealing with receiving waters of excellent quality. Moreover, it is somewhat unexpected to find that indirect emissions from the production of chemicals, pipelines and energy give such high potential impacts regarding freshwater eutrophication elsewhere (i.e. not locally) in the overall system. Another and more recently common environmental priority in the water sector is climate change mitigation, and the search for solutions to minimise system-wide greenhouse gas emissions. Our results strongly support a focus beyond that of climate change, which represent only a small part of the total life - cycle environmental impact.

Second, LCA results clearly point towards what are the environmentally most important parts and resource inputs of the system. In the case of Trondheim, treatment plants for water (VIVA) and wastewater (HØRA and LARA) together represent 66% of the total environmental impact, with the majority on the wastewater treatment side. These findings are in line with those from other studies mentioned earlier in this paper. When examining the environmental impact contributions from different resource inputs, see Figure 6, the clearly important ones are chemicals, pipeline materials and energy. Also this is in line with findings from other studies mentioned earlier. A significant reduction in environmental impact for the urban water and wastewater system in Trondheim would, theoretically, only be possible by a reduction and/or a shift in use of chemicals and energy. None of these alternatives are likely in reality, since the use of chemicals and energy is already optimised according to cost-benefit criteria in the treatment plants and since Norway already has a low-carbon electricity mix. This situation may be rather different for cities in other countries, with a more carbon-intensive electricity generation system and with less robust receiving waters. Hence, in such situations it may be important to optimise the urban water and wastewater system, from a total environmental impact perspective, by performing a trade-off between how much pollution is discharged to the receiving waters, what type of and how much chemicals are used, and how much net energy is consumed after taking into account also the possibilities of energy recovery from wastewater and sludge treatment. The LCA method would provide needed inputs to such a trade-off process.

Uncertainty

Dealing with uncertainty is a necessary part of using LCA to help model systems. Many factors can affect the results of an LCA, such as choice of inventory (LCI) and impact assessment (LCIA) methodology, system boundaries, and processes within the system. There can also be uncertainties in the parameters and assumptions included in the assessment. Energy use is an important contributor to the total impact of the system, and the choice of electricity mix will therefore have some influence on the environmental impact. The Nordic electricity mix (NORDEL), used in this study is a fairly 'clean' electricity mix due to its high share of hydropower. In order to test the sensitivity of the results to the choice of the electricity mix, the electricity mix for Central -Europe (CENTREL) was also considered. When compared to the NORDEL mix alone, the inclusion of CENTREL doubled the impacts of climate change, photochemical oxidant formation, particulate matter formation, terrestrial acidification, and fossil resource depletion (Figure 7). The total impact in these categories was still small, however, vis-à-vis the average annual impact per person. Of particular interest is that the impact of freshwater eutrophication was found to be almost ten times higher with the use of the CENTREL electricity mix, due to its higher share of electricity generated from coal. This is caused by runoff from surface landfilling of spoil and tailings from coal and lignite mining. In terms of freshwater eutrophication, the LCA results were therefore very sensitive to the electricity mix.

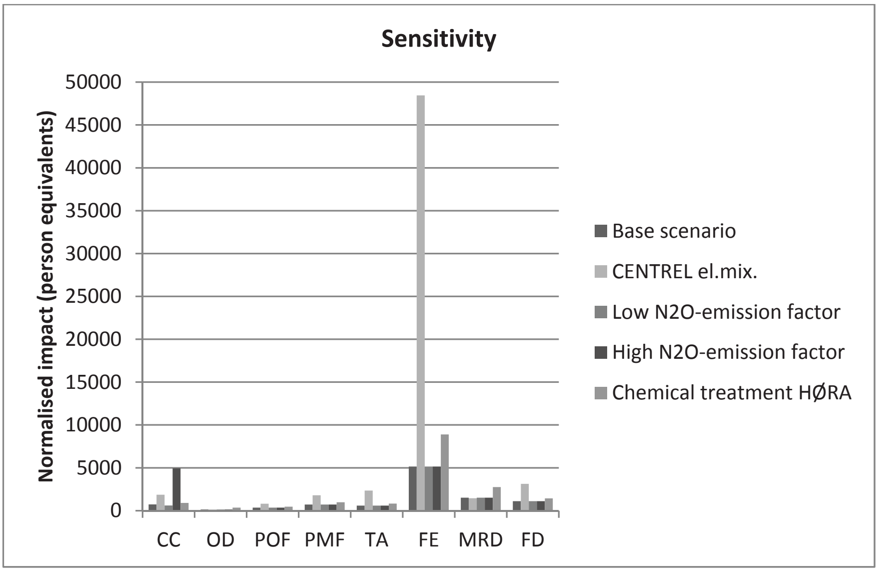

Figure 7. Sensitivity to change in electricity mix, N_2O-emission factor and wastewater treatment. Climate change (CC), ozone depletion (OD), photochemical oxidant formation (POF), particulate matter formation (PMF), terrestrial acidification (TA), freshwater eutrophication (FE), mineral resource depletion (MRD) and fossil resource depletion (FD).

Nitrous oxide contributes to 18% of the climate change impact category; however, great uncertainties are involved when calculating the climate change impact of nitrogen in the effluent. This is both due to the lack of accurate data on nitrogen content in the effluent and the uncertainty stated by the IPCC on the emission factor of N_2O (IPCC, 2006). IPCC estimates the emission factor for N_2O to be within the range of 0.0005 to 0.25. The consequences of change in emission factors were calculated in accordance with the IPCC uncertainty range. A change in emission factor from 0.005 to 0.0005 had minor influence on the results, however, a change to 0.25 increased the total impact on climate change by more than six times. Bange (2006) discussed nitrous oxide emissions in European coastal waters, and he concluded that estuaries and fjords have large emissions of N_2O to atmosphere, while open coastal areas were close to in equilibrium with the atmosphere. The situation in the Trondheimsfjord is not estimated, however.

The wastewater treatment plant HØRA today does not fully meet the requirements for removal of suspended solids, and it has therefore been discussed to introduce chemical treatment at this plant. If this is implemented, using iron chloride, the impact on ozone depletion, freshwater eutrophication, and metal depletion would increase with between 70% and 165%. The impact on climate change would also increase, but not in the same range. An increased level of wastewater treatment (i.e. aiming for higher removal efficiencies) will in most cases consume more energy and chemicals, and as a consequence increase the impact on climate change. This is an example of practical trade-offs, where the water utilities have to decide what are the most important objectives in their environmental policy.

Situations of poor raw water quality, water scarcity and sensitive receiving waters would change the choice of technologies in the water and wastewater system a lot, and thereby also the results from an LCA study. Poor water quality would imply more extensive water treatment. Water scarcity would give more attention to water savings, use of alternative water sources and pipe rehabilitation in order to avoid water leakages. Discharge to a sensitive freshwater lake/river would require more extensive wastewater treatment for extended phosphorus removal, while a sensitive fjord would require introduction of nitrification/denitrification for improved nitrogen removal.

Conclusion

The objective of this study was to examine the system-wide life cycle environmental impact potentials of operating the water and wastewater system in Trondheim, in order to clarify the relative importance of different environmental impact categories and how different elements of the water and wastewater system contribute to these impacts. The assessment provided good insight into the relative importance of different environmental impact categories, and what parts of the system and which resource inputs to the system contributed the most to each impact category, and to the total environmental impact.

The following conclusions could be drawn:

- The contributions to climate change from the water and wastewater system in Trondheim is of minor concern, compared to the total annual per capita greenhouse gas emissions. With a total impact of 8.2 million kg CO_2-eq annually or 48 kg CO_2-eq per person, this is less than 1% of a person's annual impact on climate change.
- The wastewater treatment plants contributed most (54%) to the total impact on climate change.
- Freshwater eutrophication, due to the consumption of energy and chemicals, was the environmental impact category with largest relative importance. For

the case of Trondheim this is mainly a result of indirect emissions elsewhere in the Nordic and/or global system, from electricity generation, including electricity for the production of chemicals.

- The assumptions related to the electricity mix have a strong influence on life cycle freshwater eutrophication. A shift from a Nordic electricity mix to a European electricity mix increased this category to more than ten times its original value. This large increase is mainly a result of nutrient runoff from landfilling of spoil and tailings from coal and lignite mining. Such a shift to a more dirty electricity mix would also give higher climate change impacts.

Local conditions are obviously very important in some LCA studies, and the ReCiPe model with general characterisation factors does not reflect these conditions faithfully. For the case presented in this paper, marine eutrophication was considered to represent a minor problem for emissions into a local seawater fjord, despite the results derived from the LCA calculations. This, together with great uncertainty in N_2O -emissions, was a central challenge when interpreting the LCA results within a local policy framework for future wastewater treatment strategies in Trondheim.

In the planning of a new carbon -neutral urban settlement in Trondheim, the results of this study indicate that the existing water and wastewater system is low in climate change impacts, and such a new urban settlement should rather look for greenhouse gas emission reductions outside the water sector. However, if urban water utilities in general wish to minimise their impact on climate change, they should prioritise the optimisation of chemical and energy usage, mainly in wastewater and water treatment plants. This would have to be done in a trade-off with respect to a change in treatment efficiencies and discharge of pollutants to the receiving water bodies.

Water and wastewater systems in different cities will be subject to different local conditions regarding raw water quality, water scarcity and the robustness of receiving waters. Water utilities in different cities will therefore have to face different environmental challenges and priorities. LCA can be used to identify what are the most important impact categories within the system, where in the system these impacts are created and what are their sources. All this is vital information when urban water utilities need to understand how to improve the environmental performance of their services.

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Appendix

Processes and impact factors included in the assessment:

	Process in Simapro	Changes made to processes
B1	Water works/CH/I U	
B2	Pump station CH/I U	
B3	Water storage CH/I U	
B4	Wastewater treatment plant class 2/CH/I U	
C1	Carbon dioxide, liqiud at plant/RER U	Electricity: NORDEL
C2	Sodium chloride, powder, at plant/RER U	Electricity: NORDEL
C3	EDTA, ethylenediaminetetraacetic acid, at plant/RER U	Electricity: NORDEL
C4	Iron (III) chloride, 40% in H2O, at plant/CH U	Electricity: NORDEL
D1	Diesel burned in building machine	
E1	Electricity, medium voltage, production NORDEL, at grid/NORDEL U	
E2	Light fuel oil burned at boiler, non-modulating 100 kW/CH U	Electricity: NORDEL
E3	Heat, at cogen with biogas engine, allocation exergy/CH U	Biogas production removed
E4	Heat, light fuel oil, at boiler 100 kW, non-modulating/CH U	
E5	Heat softwood logs, at wood heater 6 kW/CH U	
E6	Natural gas burned in gas turbine/GLO U	
E7	Diesel, at regional storage/RER U	
F1	Single superphosphate, as P2O5, at regional storage/RER U	
F2	Urea, as N, at regional storage/RER U	
M1	Silica sand, at plant/DE U	
P1	Polyvinchloride, at regional storage/RER U	Extrusion and transportation added
P2	Poluethylene, granulate at plant/RER U	Extrusion and transportation added
P3	Steel, low alloyed, at plant/RER U	Drawing of pipes added, scrap content and transport adjusted
P4	Cast iron at plant/RER/U	Metal product manufacturing added, scrap content and transport adjusted
P5	Concrete blocks at plant/DE U	Electricity: NORDEL, transport added
P6	Copper product manufacturing, average metal working/RER U	Transport added
P7	Glass fibre, at plant/RER U	Extrusion and transportation added
T1	Transport, lorry >32t, EURO5/RER U	
T2	Transport, lorry 3.5-7.5t, EURO5/RER U	
T3	Transport barge/RER U	
X1	Blasting/RER U	
	Other characterization factors included	
R1	Phosphorous contribution to marine eutrophication	
R2	Nitrogen contribution to marine eutrophication	
R3	N_2O contribtution to global warming	

Processes included in the assessment	**VIVA**	**Water pumps and storage**	**Water pipes**	**Wastewater pumps**	**Wastewater pipes**	**HØRA**	**LARA**
Energy	E1	E1		E1		E1, E2, E3	E1, E3, -E1, -E4, -E5, -E6
Pipes			P1, P2, P3, P4, P5, P6, P7, D1		P1, P2, P4, P5, P7, D1		
Buildings and equipment	B1	B2, B3		B2		B4, X1, E7, T1	B4, X1, E7, T1
Chemicals and material	C1, C2, M1					C3	C3, C4
Transportation of chemicals and material	T1, T2, T3					T1, T2, T3	T1, T2
Nitrogen/phosphorous						R1, R2, R3	R1, R2, R3
Fertilizer						-F1, -F2	-F1, -F2

Paper 5

Use of LCA to evaluate solutions for water and waste infrastructure in the early planning phase of carbon-neutral urban settlements.

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Use of LCA to evaluate solutions for water and waste infrastructure in the early planning phase of carbon-neutral urban settlements

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Abstract
Purpose - The paper discusses how life cycle assessment can be used in the early stage planning phase of new settlements.
Design/methodology/approach - By applying the life cycle methodology on the waste, water and wastewater system of a new carbon-neutral settlement under planning in Norway, we discuss the pros and cons with applying this methodology in an early planning phase.
Findings - The LCA methodology enabled us to compare suggestions from interdisciplinary planning teams, relate them to the existing systems in Trondheim and provide quantitative results back to the decision-makers, in this case the municipality. The environmental benefits of implementing alternative solutions in the waste, water and wastewater systems were found to be small.
Research implications/limitations - Data availability and uncertainty can be limitations in the early planning phase.
Practical implications - By applying this methodology, the life cycle environmental impact of different solutions can be assessed at an early planning stage.
Originality/value - Even if life cycle assessment has been used for years in the research community, there is too little experience with applying the methodology in the early planning phase of new projects. This paper discuss how life cycle assessment can be used to compare suggestions from interdisciplinary planning teams, relate them to existing systems and provide quantitative results back to the decision-makers.
Keywords Physical planning, Life cycle assessment, Household waste, Wastewater

Introduction
The link between greenhouse gas emissions, rising temperatures and changes to the global climate has been described by the United Nation's Intergovernmental Panel on Climate Change (IPCC, 2007). Because of this link, and the possible serious consequences of a rapidly changing climate, greenhouse gas emissions have become one of the main focuses for sustainable development. Reducing greenhouse gas emissions is challenging, and involves complex economical, ecological and social systems.

Half of the world's population live in cities, and 80 % of the world's greenhouse gas emissions are assumed to relate to urban residents and their associated affluence (Hoornweg et al., 2011). Several attempts have been done to identify where in the urban system environmental impact origin; Erickson et al. (2012) developed a consumption based model for the state of Oregon and Jones and Kammen (2011) did the same for a variety of American households. Hertwich (2011) reviewed literature estimating the life cycle impact of consumption. He identified housing and food to be important consumption categories all over the world, and mobility and purchasing of manufactured goods to be important categories in rich countries.

Because of this variety in factors important for the overall impact of living, several approaches have to be taken to reach for sustainable living. One of these approaches is to optimise the infrastructure that is added to the urban environment. Urban infrastructure is an important premise for how cities functions, and today's urban infrastructure is the result of centuries of development. In most industrialized countries we have an efficient and highly developed urban infrastructure for the treatment of solid waste, water and wastewater. These systems, however, draw on resources in the form of energy and material, process resources as nutrients and energy, and give direct and indirect emissions from treatment and disposal. In the literature there is therefore a rising concern that today's infrastructure systems are unsustainable; Guest et al. (2009) and Larsen et al. (2009b) ask for a paradigm shift in wastewater handling, Astrup (2011) stresses the importance of seeing waste as a resource, and Agudelo-Vera et al. (2011) claim that bringing resource management into urban planning is one of the most important steps towards sustainable urban planning.

When planning the new urban carbon-neutral settlement at Brøset in Trondheim, one of the aims was to choose infrastructure solutions for waste, water and wastewater with the minimum emissions of greenhouse gases. There was little knowledge about how to do that, which solutions to consider, and possible levels of emission reductions. The urban planners, design teams and stakeholders involved in the early planning at Brøset were questioning how to get more quantitative knowledge, as a basis for zooming in on selected design solutions. They were also questioning the role and relative importance of these infrastructure systems in relation to the overall impact of an ambitious carbon-neutral urban settlement project.

New construction will always cause emissions. Heinonen et al. (2012) question whether new, low-impact settlements should be seen as a good strategy for reducing environmental impact at a city level. They show the importance of the impact of the construction phase when building new settlements, and they conclude than one should compare the life cycle impact of different solutions. There is, however, no clear place for life cycle environmental objectives in the planning phase of a settlement, because the stakeholders (such as the municipality, the developers or the future inhabitants) usually have no special interest in achieving low environmental impact

(Wallbaum et al., 2011). The urban carbon-neutral settlement project at Brøset, in Trondheim, is an exception to this. In this case the municipality asked for a cooperation with researchers in order to include environmental assessments more actively in the decision-support process.

The case of planning a carbon-neutral urban settlement at Brøset

Brøset is a planned new, research-initiated settlement in Trondheim, Norway, with carbon-neutral ambitions. The 35-hectare (350,000 square metres) site is situated 4 km from the city centre, and is a greenfield area in a suburban environment. There are now plans for approximately 1600 dwellings, with an estimated 3500 inhabitants; in addition a school, a large kindergarten and small-scale businesses are included in the plan. The municipality of Trondheim has included this project as one of its contributions to the national 'Future cities' project run by the Norwegian Ministry of the Environment. A researcher team has been closely involved in the planning process at Brøset, examining processes, concept development and implementation of carbon-neutral solutions (Figure 1).

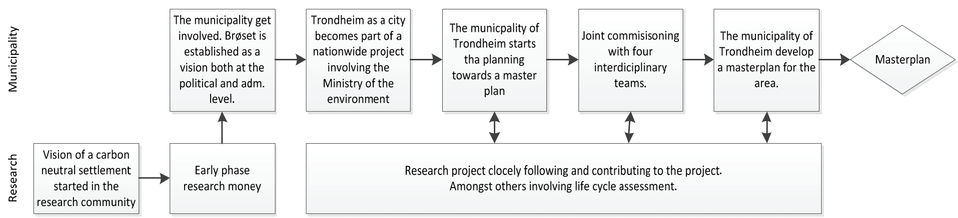

Figure 1. Planning process at Brøset.

As cooperation and open dialogue has been very important in the progress of the planning, it was decided that a 'joint commissioning process' for the master-plan development would be undertaken. This was a process whereby design teams pre-qualified to be a part of the project. Selected teams then participated in joint workshops, with each team developing individual suggestions for a master plan at Brøset, but without the intention of choosing a winning team. The advantage of this type of process instead of a conventional competition was argued to be that the pre-qualified teams could share experiences, that the process could be better influenced, and that by not having a winning team all the suggestions could be used as inspiration for the final master plan developed by the municipality. Four Nordic interdisciplinary teams participated in the commissioning.

At a very early stage, before the joint commissioning, a consumption-based input-output analysis (IOA) methodology was used to estimate the average Norwegian per capita impact on global warming, with the aim to create a vision for the new settlement (Solli et al., 2010). Using this, and the IPCC scenario of a projected 2 °C increase in global temperature, an objective was set that the average Brøset inhabitant should be responsible for greenhouse emissions 75 % lower than the national average.

The vision for the area is thereby: '*Brøset as a future-oriented, attractive and climate neutral neighbourhood with less than 3 tonnes CO_2-eq per capita*'. It was seen as impossible to achieve a complete carbon-neutral level at this stage.

The main objectives for the new settlement were laid out in the programme for the joint commissioning process (Trondheim kommune, 2010), as follows:

- Reduce (consumption-based) CO2-eq emissions by 60–90 % compared to similar urban density areas;
- Establish an ecologically sustainable city environment with high architectural quality;
- Optimise high density with quality, functions and costs in order to secure a sound social economy;
- Give the area a physical design that allows for and inspires a climate friendly lifestyle;
- Arrange for user-participation to secure the needed evolvement of the area over time.

The design teams presented a series of alternatives for how Brøset could be developed, regarding urban density and form, green areas, design of buildings incl. use of passive house technologies, and solutions for transportation, energy, water, wastewater and solid waste. In this article we only examine the design teams' suggestions related to infrastructure for water, wastewater and waste. These suggestions were, not surprisingly, at such an early stage of planning, somewhat vague concerning technical solutions and infrastructure design, but the most important elements of what was proposed can be seen underneath.

Team A:

- Waste prevention due to access to free stores, equipment library, markets, reuse centre
- Keeping animals at the site, which can be fed with food waste
- Underground waste collection
- Compost and use of CO_2 in greenhouses
- Food production at the site – more expensive, but less food waste
- Good provision for waste sorting
- Local stormwater handling with green roofs and walls, retention, permeable covering, local treatment of wastewater in, for example, raingardens.

Team B:

- Waste prevention due to less consumption, local food production, equipment library, reuse centre, free store
- Composting in summer, biogas production in winter, CO_2 storage in the soil
- Local stormwater treatment with green roofs, water retention, constructed wetland, infiltration etc.

Team C:

- Waste prevention due to local food production, equipment library, re-design
- More extensive recycling
- Underground waste collection
- Extended producer responsibility for waste
- Local stormwater treatment with retention, infiltration, green walls and local treatment

Team D:

- Waste prevention due to local food production, equipment library
- Composting or local biogas production
- Local stormwater treatment with retention, infiltration, green roofs and walls etc.
- Separation of greywater and blackwater, with recycling of greywater, and blackwater sent to a centralised wastewater treatment plant with sun-dried slag

The suggestions from the design teams listed above are intended to contribute to the goals outlined in the planning programme, with attention on waste prevention, underground waste systems, good availability for recycling and local stormwater treatment. In addition composting, CO_2-use in greenhouses, biogas production, introducing animals for utilisation of food waste and separation of greywater and blackwater are suggested.

In the city of Trondheim the waste system is based on recycling and incineration with heat recovery, where the heat recovered is the main heating source for a district heating system. Only 3 % of the city's waste is landfilled. Water is taken from a lake, treated and piped to the consumers with a water loss of 32 %. The wastewater from households, businesses and industry is collected, partly together with stormwater, and sent to two local treatment plants. The processes used at the wastewater treatment plants consist of phosphorous removal steps, either mechanical or chemical, and biogas collection including utilisation. The least contaminated sludge is used in agriculture.

Methodology

Scenario development

Based on the suggestions from the interdisciplinary design teams, our challenge was to translate these into a selection of more specific technological solutions, which we could further examine in terms of environmental quality, with the use of life cycle assessment (LCA) methods, at a comparable level. Some of the suggestions were omitted, such as composting, extended producer responsibility, the keeping of animals fed on food waste, and underground vacuum systems for waste collection. Composting does not utilise the embodied energy in food waste in the same way as incineration with heat recovery or anaerobic digestion with heat production, and is only recommended in special situations (Finnveden et al., 2005). Producer responsibility was regarded outside the scope of the present study, since this is implemented more on a regional/national scale. Keeping animals at the site was seen as unrealistic due to the level of density planned for the area. Underground vacuum systems were not tested explicitly in this study, but Iriarte et al. (2009) found them to be more emission intensive than container systems due to production of pipes and energy-use during operation. Such systems will still be included in the Brøset settlement, however, due to other advantages.

The remaining suggestions were made use of, as far as possible, and implemented (as combinations of technical solutions) in a set of scenarios for further analysis. However, as some of the design team's suggestions were somewhat vague, we also had to develop the scenarios on the basis of knowledge from assessment of business-as-usual scenarios, findings in literature, information from other similar (ambitious) urban settlements projects, and local knowledge. For the wastewater system one team suggested separation of greywater and blackwater. Assessment of the business-as-usual case for the water and wastewater system showed that the environmental impact of the current system is actually low. Few alternative decentralised systems for the treatment of wastewater would reduce the impact further. According to literature, treatment of greywater in constructed wetlands is the only alternative solution for local treatment of wastewater worth assessing in this case.

These are all examples of how we combined the suggestions from the design teams with other information, when developing a set of scenarios for LCA testing. In doing so we also had to take seriously into consideration the existing infrastructure solutions for water, wastewater and waste at the city-level in Trondheim, since a new urban settlement at Brøset, maybe with some new local solutions, is of course not completely disconnected from its surrounding infrastructure. In the end, the final set of scenarios for LCA testing was decided by ourselves as researchers in agreement with the project owner. These scenarios are listed below:

For waste infrastructure:

- Scenario 1: Business-as-usual
- Scenario 2: Source-separation of food waste, a centralised biogas plant, upgrading of biogas to fuel, the other fractions as in business-as-usual
- Scenario 3: Source-separation of food waste, local biogas plant, biogas used in a combined heat and power plant, the other fractions as in business-as-usual
- Scenario 4: Increased recycling, 90% source-separation of paper, glass and metals, 70 % separation of plastic

For water and wastewater infrastructure:
- Scenario A: Business-as-usual. Stormwater to the wastewater treatment plant (WWTP)
- Scenario B: Installation of water saving appliances. Water consumption down from 160 l/p/d to 105 l/p/d
- Scenario C: Local stormwater treatment. No stormwater to the WWTP
- Scenario D: Local grey water treatment in subsurface constructed wetlands. Stormwater to the WWTP

Life cycle assessment method

Life cycle assessment (LCA) has been used in the research communities for many years, originally to assess environmental impact of products, but later also to look at systems of different scales. The methodology is covered by ISO 14040 and ISO 14044 and follows four stages (1) goal and scope definition, (2) life cycle inventory, (3) life cycle impact assessment and (4) interpretation (ISO, 2006a, ISO, 2006b). The waste research field in particular has used this methodology extensively at all scales, carrying out assessments of waste management systems of countries and cities (Cherubini et al., 2009, Eriksson et al., 2005, Larsen et al., 2010, Raadal et al., 2009), fractions (Astrup et al., 2009, Larsen et al., 2009a, Merrild et al., 2009), and specific elements of waste management systems (Eisted et al., 2009, Rives et al., 2010). In the last decade water and wastewater systems also have been assessed using this methodology (Lassaux et al., 2007, Lundie et al., 2004, Venkatesh and Brattebo, 2011). While there are other tools available for assessing the environmental impact of different systems, the unique feature of LCA compared to methods such as Strategic Environmental Assessments, Cost-Benefit Analysis, Material Flow Analysis or Ecological Footprints is the comprehensive life-cycle perspective (Finnveden et al., 2009). The resent years a consumption-based life-cycle approach has gained increased interest. One reason is because it overcomes some constraints of LCA; LCA can be time consuming, complex and the cut-off criteria applied can have a significant impact on the results (Suh et al., 2004). One valued feature of the consumption-based models is the possibility to include the indirect impact of goods produced abroad. Inclusion of these emissions can have an significant impact on the results from a country, city or settlement assessment (Erickson et al., 2012). A consumption-based model would, however, be too aggregated for the comparison of different infrastructure systems, which is the objective of the assessment in this article.

Eriksson and Baky (2010) tested potential key parameters in system analysis of municipal solid waste management and claim that LCA results are robust. We, however, have to be aware that all methodologies for modelling and assessing real systems, including LCA, have limitations. Gentil et al. (2010) reviewed the importance of technical assumptions in models for waste management and found that the functional unit, system boundaries, waste composition and energy modelling all have significant impact on the results. Bras-Klapwijk (1998) discuss how, when used in policy making, LCA can be misused due to the apparent objectivity and the quantitative nature of the results. The same is discussed by Lazarevic et al. (2012) when looking at the application of life-cycle thinking in European waste policy. They are concerned how different stakeholder involvement can be used to assess LCA in a polarising way. It is important to be aware of these possible constraints when applying LCA methodology.

In this study, LCA was used to investigate if alternative centralised or decentralised solutions could decrease the environmental impact from the infrastructure systems involved. The main environmental objective for the urban settlement at Brøset is to become close to carbon-neutral, however, any solution for water, wastewater and waste infrastructure will of course also lead to other environmental impacts, and there could be trade-offs between the different types of impacts. To be aware of these possible trade-offs several impact categories were included in the assessments.

System boundaries

Defining system boundaries is one of the first steps of performing an LCA, and a crucial one. When performing LCA on products we usually follow the product from cradle-to-grave or cradle-to-gate. However, when modelling a waste management system, waste is followed from its entering the waste system to its end destination, whether this is landfill, recycling or energy recovery (Figure 2). This is often called a "no-burden" approach, where the impact related to the production and use phase of the products that eventually become waste is not included in the system. If we wanted to include upstream impacts from the production and use phase, the LCA method would become extremely complex, and a better strategy would be to use a combined IO-LCA (input-output LCA) method. The problem when using the no-burden approach is, however, that it is not possible to include waste prevention (introducing less waste into the waste system) in the calculations. The functional unit that is used for the waste system is "collection, transport and treatment during one year, of the waste streams of mixed waste, paper, plastic, glass and metals from 1500 new households (3315 persons) at Brøset in Trondheim, Norway".

In the waste management system resources in the waste is utilised to substitute other energy or material sources. To include this in the calculations we estimated the amount of substituted energy and materials and their potential environmental impact. Mixed waste is, for example, used in the incinerator to produce heat for the district heating system in Trondheim. Since the incinerator is the main energy source in the district heating system, we assumed that other heating sources, mainly based on electricity, were substituted. Peak demand energy sources in the district heating system were not included, as we assumed that including the waste from Brøset would have little effect on the total balance of the district heating system.

For the recycled materials we assumed replacement of the production of virgin materials. Loss in material quality was accounted for by replacing less than 100 % of the virgin material. Substitution is important for the outcome of the assessment, often resulting in negative net impact from the waste system. This is because we are avoiding landfill, and the energy retrieved from the incinerator is valuable in Norway's cold climate, despite the widespread access to fairly clean energy from hydropower.

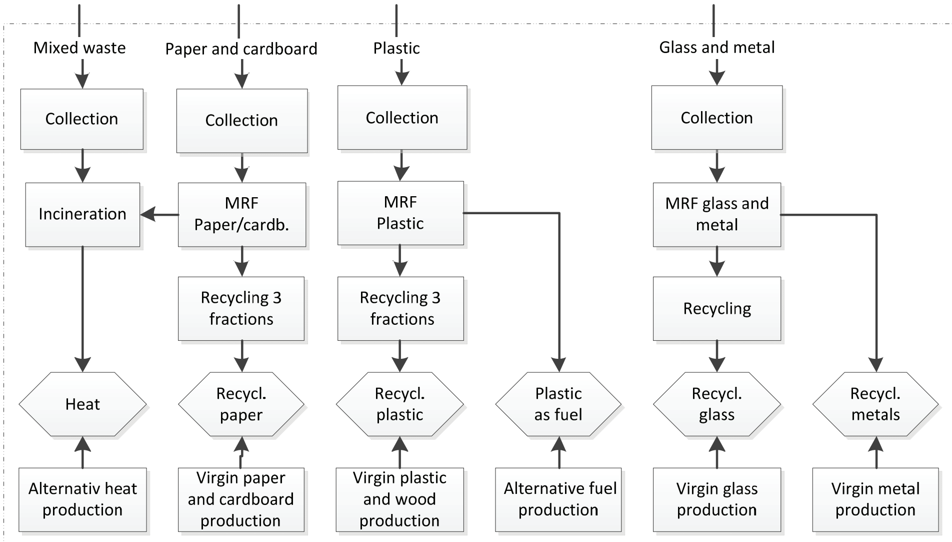

Figure 2. The business-as-usual waste system. Transportation is included.

For the water and wastewater system we included impacts occurring from the water treatment plant (WTP) through to discharge in seawater (Figure 3). The functional unit was "one year provision of water and collection, transportation and treatment of wastewater (including stormwater) for the Brøset settlement (3500 persons) in Trondheim, Norway". Brøset will be connected to a mechanical-chemical wastewater treatment plant (WWTP) with phosphorous removal. Infrastructure was included for the large-scale system, but not infrastructure inside the new settlement itself, such as pipelines, construction of stormwater systems etc. The building structure and density had not been decided at the time of the assessment, and it was therefore considered too early in the planning process to get good estimates of the amounts of different materials needed. Substitution of energy due to heat utilisation of biogas and substitution of mineral fertilizer due to the fertilizer value of the sludge were accounted for.

To perform the LCAs we used two different LCA tools. Easewaste 2008 (Kirkeby et al., 2006), developed at the Technical University of Denmark, is a designated LCA tool for waste systems. It includes the EDIP 97 impact assessment method, where the environmental impacts are normalised according to EDIP97 values of global or EU-15 annual environmental impacts of one person. The results are given in person equivalents (PE). Simapro, the general LCA tool (Pré Consultants, 2011), was used for the assessment of the water and wastewater systems. Simapro incorporates multiple databases, of which the Ecoinvent database was used in the assessment of the water and wastewater system. The impact assessment method applied was ReCiPe midpoint (H) v1.06 (July 2011) and the impacts were normalised against average annual impact per person in Europe, and given in PE. The number of inhabitants in the models was 3315 for the waste system and 3500 for the water and wastewater system.

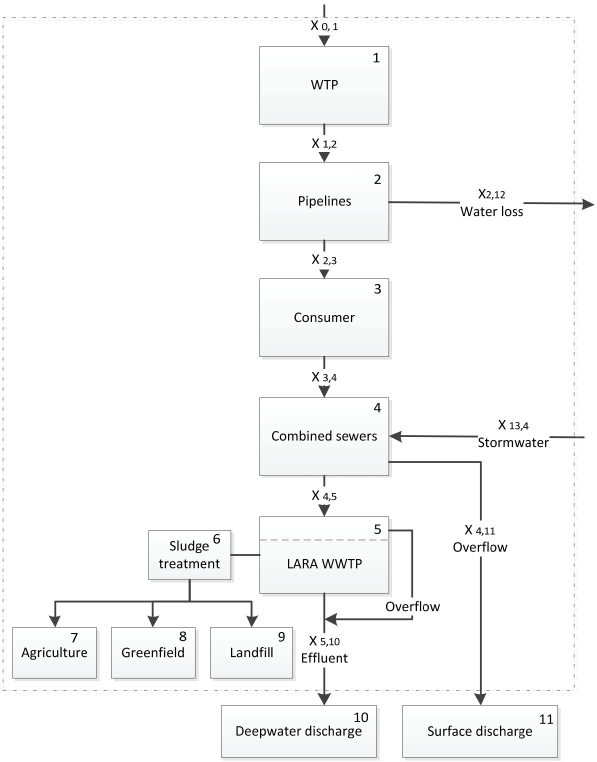

Figure 3. Business-as-usual water and wastewater system.

Results and discussion

The results from the assessment of the alternative solutions for solid waste show that the impacts of source separation and digestion of food waste did not differ significantly from today's practice of incineration. Brøset should therefore rather connect to the conventional (existing) infrastructure system. At the same time the results show that if phosphorous scarcity becomes a problem in the future, as discussed by Neset et al. (2012), there is a potential for anaerobic digestion of food waste with recovery of phosphorous from the digestives without increasing the impact on climate change. Food waste separation and treatment can be implemented in both small and large scale systems, and can be combined with source separation and treatment of blackwater as seen in Flintenbreite in Germany (GTZ Ecosan project, 2005). While we found a net saving to the environment in most impact categories for the waste system, the water and wastewater system had a net load in all categories. The impact on global warming was, however, less than 1 % of annual global warming impact per person. Both installing water saving appliances and treatment of stormwater locally would reduce the total impact from the system. More details are given underneath.

The waste system

The results from assessing all waste scenarios show that increased source-separation of paper, plastic, glass and metals (Scenario 4) is beneficial for the global warming and ecotoxicity in water impact categories (Figure 4). There are, however, trade-offs; the impact on human toxicity becomes less beneficial, due to less replaced electricity production from the incineration process. Introducing source-separation and digestion of food waste (Scenario 2 and 3) was found to have similar impacts to the current practice of incinerating food waste. Transportation was found to have little importance in all the scenarios, due to short distances. The total impact from the waste system is negative for global warming, indicating that the waste system avoids environmental impact by replacing production of virgin materials and energy; especially important is replacement of virgin paper production. More details on the scenarios and results of the waste system can be found in Slagstad and Brattebø (2012).

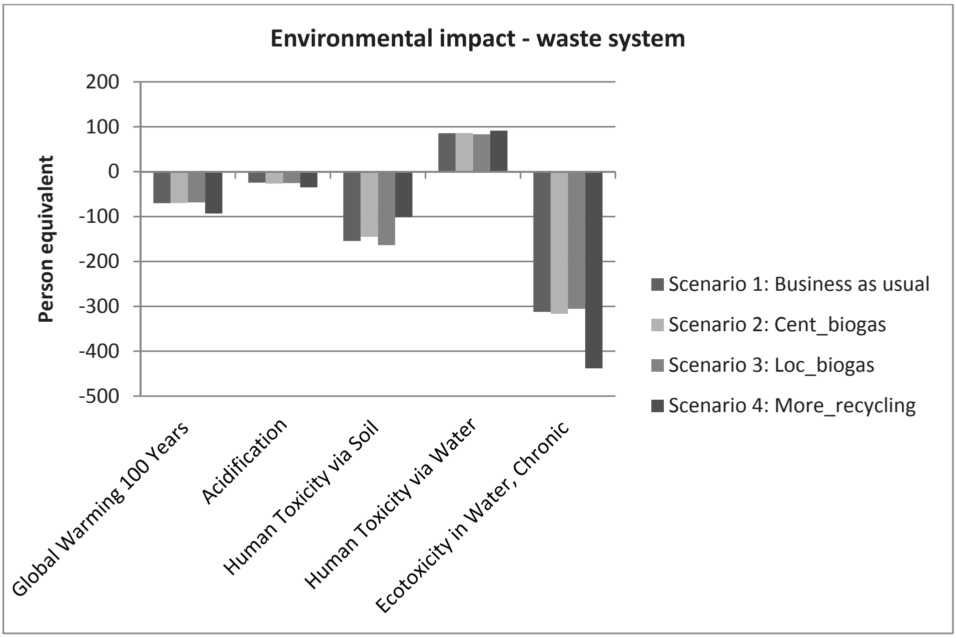

Figure 4. Environmental impact from the waste system. Relates to 3315 persons.

The water and wastewater system

When assessing the scenarios for the water and wastewater system we found that two scenarios, B and C, had improvements in all impact categories compared to business-as-usual (Figure 5). The improvements are, however, fairly small. Applying both Scenario B and C would reduce the impact on climate change of the water and wastewater system from 36 to 22 kg CO_2-eq per person annually. Introducing constructed wetland has trade-offs between the impact categories: some impacts increasing compared to the business-as-usual scenario, and some decreasing. None of the scenarios achieves carbon-neutrality.

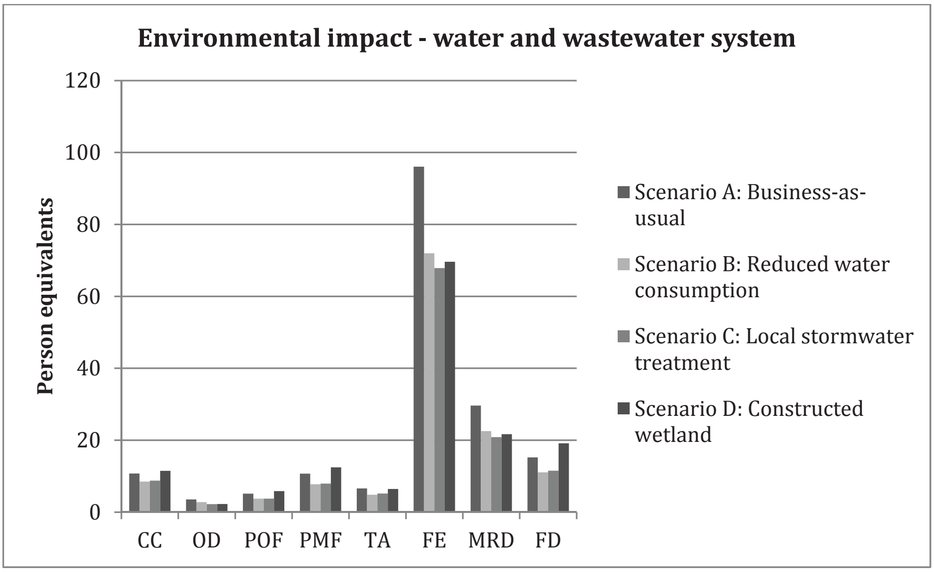

Figure 5. Environmental impact from the water and wastewater system. Related to 3500 people. Climate change (CC), Ozone depletion (OD), Photochemical oxidant formation (POF), Particulate matter formation (PMF), Terrestrial acidification (TA), Freshwater eutrophication (FE), Mineral resource depletion (MRD) and Fossil resource depletion (FD).

The municipality was informed about the results of the assessment, and local solutions for stormwater treatment and targets for water saving for the new settlement were suggested (together with waste prevention targets). Since the difference between the scenarios in overall impact were small, it was difficult to evaluate the influence the results had on decision making in the municipality. The master plan is now nearly complete, local wastewater treatment is implemented together with waste prevention targets. The municipality decided not to include water saving targets at the master plan level. No other stakeholders were involved at this stage.

LCA and uncertainty

The fact that this study was carried out during the early planning phase adds uncertainty to the results. We had to estimate input parameters such as the waste composition, sorting efficiencies, water use and wastewater production. These input parameters were estimated based on either average Trondheim data or on studies of similar density areas within Trondheim. What we do not know, and what is very difficult to estimate, is the differences between this ambitious project and conventional neighbourhoods. The Brøset area will most probably attract people more willing to adapt to a low-impact society, and we might see changes in inhabitant behaviour. A change in behaviour can, for example, affect the waste composition, which is important for the results of the assessment. Life cycle assessment can at best model the present, and estimate the future. When using LCA in to support policy making, this can be a constraint for a sustainable development, as discussed by Lazarevic et al. (2012).

When we compare alternative solutions, some of them theoretical, the availability of data for the technology is important. For the alternative solutions, we had to base our analyses on commercial databases or literature data, and hence, our assumptions are not adjusted to local conditions. For the water and wastewater system the estimation of emissions of N_2O, a potent greenhouse gas, was the most challenging issue, as we had no measurements available of the amount of nitrogen in the wastewater effluent to the receiving waters. This problem is, however, not limited to the early planning phase we were examining, but a more general problem when assessing the wastewater system of Trondheim, and thereby an example of the problem of data availability when performing LCA in all phases of projects. Based on the level of uncertainty in the assessments it would be interesting to compare the findings in the assessments with the actual performance of the solutions chosen.

LCA in planning

Opportunities in using the life cycle methodology are many. LCA offers help in expanding our perspectives; we get a good overview both of the total impact from a system and where in the system the main contributors to the impact can be found and we can consider trade-offs between environmental impact categories when comparing alternative systems. LCA does not take only direct emissions into account, but also includes impacts resulting from production and transportation of resources, construction and maintenance of buildings and infrastructure, and end-of life management. Ekvall et al. (2007) evaluated the use of LCA in waste management research and explained how important the indirect environmental impacts can be for the total impact from the system.

LCA has been used to assess alternative technical solutions; however, the results of the assessments show the importance of inhabitant behaviour, with increased source separation and reduced water consumption as two sources for reduced impact on global warming. There are technical appliances that to some extent can reduce water consumption, however, the inhabitants' water-use behaviour would have to change if the consumption level we have indicated in this assessment was to be achieved. The results are therefore based on inhabitants being willing and able to change their behaviour. However, residents are not participating in the early phase planning at this detailing level. To make allowances for this, LCA could be combined with indicator-based sustainability assessments, with global warming as one of the indicators. In this way social implications of measures considered could be valued together with economic and environmental indicators. Weighting would also need to be introduced. In the case presented in this paper there were few stakeholders involved in the early phase and the environmental goals for the area, focusing on carbon-neutrality, were clear. In addition one of the conclusions from the evaluation of ambitious environmental planning projects in Norway was that there should not be too many environmental objectives (Narvestad, 2010). Life cycle assessment was therefore found to be the best alternative for assessing the impact from infrastructure at this stage in the process.

In the assessment of this case study, the waste, water and wastewater system was found to have little influence on the total environmental impact of the new settlement. Although there were few technical solutions potentially improving the impact from the system, the LCA identified that consumption of energy and chemicals in the wastewater treatment plants are important contributors to the total impact from the wastewater system in Trondheim. Optimisation of these processes could therefore be more important than new technical solutions at the neighbourhood level.

The case study included in this article is only an example of how LCA can be included in the planning of new settlements; impacts from other parts of the project, such as transportation, energy use and buildings could also be evaluated with the help of LCA. Although LCA has been an important contributor to understanding the role of the waste, water and wastewater system, and to comparing the conventional system with alternative systems in the Brøset project, future planning processes of settlements in urban areas with existing well-developed infrastructure for waste, water and wastewater might concentrate on other opportunities for minimizing carbon emissions. Important contributors to carbon emissions could be identified by Environmentally Extended Input-Output analyses before performing the LCA, in this way LCA still would be useful in the early planning phase, but the time and investment put down in performing LCA could be confined to systems where an improvement in system environmental impact has a larger effect.

Conclusion

LCA has shown to be useful for the evaluation of suggested water, wastewater and waste infrastructure solutions for a new urban settlement, in the early planning phase of the project. This was because this was a new kind of project with very high ambitions and with little knowledge about the importance of alternative infrastructure systems and how they should be quantitatively compared at this stage. There are some general limitations to LCA methodology, and some added uncertainty, when applying LCA so early in the planning process. Nevertheless, the methodology enabled us to compare suggestions from interdisciplinary planning teams, relate them to the existing systems in Trondheim and provide quantitative results back to the municipality being responsible for decision-making in the further planning process. For the waste, water and wastewater systems in our case study, it was shown that a focus on recycling, water savings and local stormwater treatment would give the most important opportunities for reducing the new urban settlement's impact on global warming, however, these are all small contributions to the overall reduction in impact possible for such a settlement. Identifying the most important contributors to global warming before deciding on which systems to analyse would increase the possible impact of applying LCA in the early planning phase.

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Paper 6

Environmental impact of water, wastewater and waste infrastructure.

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Draft for a book chapter

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