NTNU – Trondheim
Norwegian University of
Science and Technology

Modelling environmental benefits of household waste prevention

Eléonore Lèbre

Master in Industrial Ecology
Submission date: June 2012
Supervisor: Helge Brattebø, IVM
Co-supervisor: Helene Slagstad, IVM

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

Preface

The present report has been developed as a master thesis in Industrial Ecology at the Department of Hydraulic and Environmental Engineering at the Norwegian University of Science and Technology under the supervision of Helge Brattebø and Helene Slagstad. The study started in January 2012 and accounts for a total of 30 ECTS. It follows a previous project done on the same topic in the fall semester 2011: *Life cycle assessment of waste prevention activities: a review of methodologies.* This project's goal was to review the different methodologies found in the literature that were able to assess the environmental impacts of waste prevention. They were explained and compared as regard to what waste prevention is. From this project, a particular methodology was recommended for assessing waste prevention at the household level; it was applied in the present master thesis.

The environmental benefits of sustainable consumption and waste prevention at the household level were of particular interest to me because I have often wondered what I should do to reduce my own environmental footprint. This master thesis offered me the opportunity to read and learn about this interesting topic that is the life cycle environmental impact of products. It showed important aspects of consumption that are often not seen by consumers themselves: the future of their waste and the indirect impacts embodied in their purchases. This topic also offered me the opportunity to work on two distinct frameworks, using two different softwares and acquiring skills on both Input Output Analysis and Life Cycle Assessment of Waste Management Systems. My interest in Life Cycle Assessment has not declined as I have been able to see the bigger picture of it, know its limitations and its alternatives in certain situations.

I would like to thank my supervisors Helge Brattebø and Helene Slagstad for their support and their guidance throughout the year. Special thanks to Helene for answering patiently all my questions about Easewaste and the waste management system and to Helge for his wise and systemic approach on the topic. I would also like to thank Edgar Hertwich for introducing me to the Exiopol Input Output models and Richard Wood, Kjartan Steen-Olsen and Anthony Pak for answering all my questions on this topic. Thank you again to Hanne Møller from Østfoldforskning for sharing her data on the food packaging. Finally I would like to address a special thanks to all the people involved in the Industrial Ecology program, students, professors and researchers, for being concerned by this important issue that is the human responsibility on its environment and for raising so many interesting debates in and outside the classroom. I thank my parents and my home university as well for supporting me into integrating this MSc program.

Eléonore Lèbre

Summary

Waste prevention can be seen as a form of waste treatment, and it is then considered as the most desirable option to mitigate the environmental impacts of waste generation. However, some have already pointed out the fact that the true potential of waste prevention might lie in its connection to sustainable consumption, and not as a substitute to waste treatment (Ekvall 2008, Olofsson 2004). Sustainable consumption and waste prevention are concepts that are closely related. Goods that people consume always end up as waste. In some cases, waste prevention also results in reduced consumption and this is what was analysed in this master thesis. The aim is to assess the environmental benefits of household waste prevention by considering the overall production chain and not only the waste management system.

To assess the benefits of waste prevention, a hybrid LCA model was developed. This model combines an Input Output Analysis of consumer expenditures with a Life Cycle Analysis of household waste generation. The Input Output Analysis is an appropriate tool to assess a basket of product categories that are expressed in monetary terms, as it is the case for household consumption. The Input Output dataset is connected to the Consumer Expenditure Survey which gathers a household's total yearly purchases. The Life Cycle Analysis of a waste generation vector includes all data characteristic of the Waste Management System in Trondheim and the future of the waste is assessed from its collection and sorting to its recycling and use in secondary production.

The scenarios chosen to evaluate the potential of waste prevention are targeting food, textiles and paper products. They all assume that waste prevention results in a proportional decrease in consumption, thus affecting both the LCA and the IOA results. The aim is to compare these two sets of results. In the IO scenarios, the influence of a rebound effect was also tested. Rebound effects are due to a constant income that settles the total amount of expenditure: scenarios generate reduced consumption and hence money savings that will still be spent on something else in the end. The ways they are re-spent will determin the final results. Various cases were tested: the rebound on holidays, restaurant, culture, repair, the marginal rebound and the simple rebound. The two latter ones distribute the savings on all categories.

From this study the main results are the following:

- The environmental benefits of waste prevention occur mostly at the production chain level: most changes occurring at the waste management level are from 0.1 to maximum 10% of the ones occurring upstream. Benefits generated by the waste management system are low and sometimes even negative, meaning that the reduction scenarios generated more impact than the reference scenario. This is because the Waste Management System generates environmental benefits on its own, thanks to energy recovery and material recycling that substitute primary production. Decreasing the amounts of waste collected hence reduces these benefits.

- The influence of the rebound effects on the results is significant. In the case of global warming, the holiday rebound is the one that mitigates the most the initial benefits (they are reduced from 7% to 1% in the results that combine all scenarios together). The marginal and the simple rebounds come next. Rebounds on restaurant and culture are most of the time the most beneficial in the way that they reduce the benefits only from 7% to 5.5%.

- Even though comparison between the different targeted categories is subject to uncertainties, one can still notice the importance of food, which generates significant benefits even though the consumption was only reduced by 11%. The results also show that preventing paper waste is the least beneficial scenario.

Table of content

List of figures

List of tables

Abbreviations

MRIO: Multi-Regional Input Output

IOA: Input Output Analysis

LCA: Life Cycle Analysis

WPA: Waste Prevention Activity

CES: Consumer Expenditure Survey

WGV: Waste Generation Vector

ROW: Rest Of the World

GHG: Greenhouse Gas

MPS: Marginal Propensity to Spend

Definitions (source: economymap.org):

GWP: Global Warming Potential
"Global Warming Potential, or greenhouse effect, is the impact of human emissions on the heat absorption of the atmosphere, which may have adverse impacts on ecosystem health, human health and material welfare. GWP is measured in kilograms of CO_2 equivalent (kg CO_2-eq)."

AP: Acidification Potential
"Acidification Potential is the result of emissions of acidifying air pollutants, such as SO_2 or NO_x, which have negative impacts on soil, groundwater, surface waters, biological organisms, ecosystems and materials. AP is measured in kg of SO_2 equivalent."

POCP: Photochemical Oxidation Potential
"It is the formation of reactive chemical compounds by the action of sunlight on certain air pollutants. These compounds may be injurious to human health, ecosystems, materials and crops. POCP is measured in kg of ethylene equivalent (kg C_2H_4-eq)."

1. Introduction

1.1. Motivation

Environmental concern has now been developing rapidly as the damages caused to the Earth are growing and getting more and more visible. Facing urgent and worrying issues such as global warming and resource depletion, we realize that it is essential to mitigate the environmental damages caused by our consumption, which is the core of the problem. Sustainable consumption and waste prevention are two important concepts in that matter, trying to solve the same big issue, one by focusing on what is purchased, and the other one on what is wasted.

Improvements have to be made at every step of a good or a service's value chain. Producers have to find cleaner ways to produce. Governments are creating laws, protocols and regulations to encourage more environmentally friendly behaviours from both companies and individuals. Households play an important role representing the consumers, the ones for whom products are made and the ones who elect the politicians. Moreover households are the second greatest producers of waste in Norway (21% of national waste generation; Statistics Norway, 2010), after manufacturing industries (26%). Average household expenditure represents more than 56% of the GDP in Europe (Eurostat, 2006). Households are thus by far the biggest consumers in developed countries, bigger than governmental expenses and capital investments. A positive change in households' behaviour could have significant environmental benefits. This was the main motivation for investigating various activities at the household level in this paper.

However, one has to be aware that sustainable consumption and waste prevention are much broader topics than what this master thesis focuses on: producers can design their products to last longer, be lighter and reparable; at the design stage, one can already act on the environmental impact that a product will have in its use phase; logistics and transportation can be optimized in order to reduce travel distances, loading volumes, packaging amounts and losses; engineers can work on optimizing the production phase by using energy efficient machinery etc. All of the above play a role in making a product less harmful to the environment and decreasing its environmental impact.

1.2. What is waste prevention?

Before going any further we should first define what waste prevention is. Salhofer et al. (2008), Cleary (2010) and Ekvall (2008) discuss what waste prevention is and how it should be dealt with. Preventing waste involves many different kinds of activities, applied at different levels through various measures, as discussed earlier. Cleary (2010) differentiates eight kinds of waste prevention activities (WPA) that summarize the practical ways in which waste prevention can occur:

The first, most basic way to reduce waste is to reduce consumption. One reduces its consumption of one good or a basket of goods without any substitution to other products, for example the reduction of advertising mails. Here the function provided by advertising mails is simply reduced along with consumption and waste generation: the service level is decreased. Other types of WPA involve dematerialization, which aim at keeping approximately the same service level with less material. Reusing a disposable good is one of them, as well as substituting a material good by a service or a capital good (reading online newspaper, for example) or again substituting a disposable good by a reusable one. Keeping the same material good, there is always the possibility of lightening it or extending its life span via a new design or a by a more careful use. Finally Cleary (2010) points out two other ways to prevent waste: on property residential waste treatment (such as backyard composting) and storage of waste products (such as keeping an old cell phone in a drawer). In the latter option the waste is only prevented in the sense that it does not enter the waste management system. However it is still generated. In all of these cases the household plays a role in the way it chooses what it purchases and in the way it makes use of the products it owns.

One can notice that all of these WPA types, except the two last ones that are actually closer to waste diversion than to actual waste prevention, involve a reduction or a switch in consumption patterns. Preventing waste leads to a decrease in material use and therefore a reduction in material production. Waste prevention has an influence on upstream processes in the way it is connected to consumption.

Hence, sustainable consumption and waste prevention are concepts that are closely related; so closely that they should not be considered separately. Goods that people consume always end up as waste; and a waste prevention activity that does not imply a change in consumption will not achieve great environmental benefits (Ekvall, 2008). Waste prevention is often seen as a form of waste treatment (Gentil 2011, Cleary 2010), and it is then considered as the most preferable option. Moreover, it was assigned the highest priority under European waste management law (Salhofer et al., 2008). However, some have already pointed out the fact that the true potential of waste prevention might lie in its connection to sustainable consumption, and not as a substitute to waste treatment (Ekvall 2008, Olofsson 2004).

1.3. Aim and scope

This paper aims at quantifying the environmental benefits of a waste prevention activity taking place at the household level in Trondheim. The system investigated can be summarized as follows:

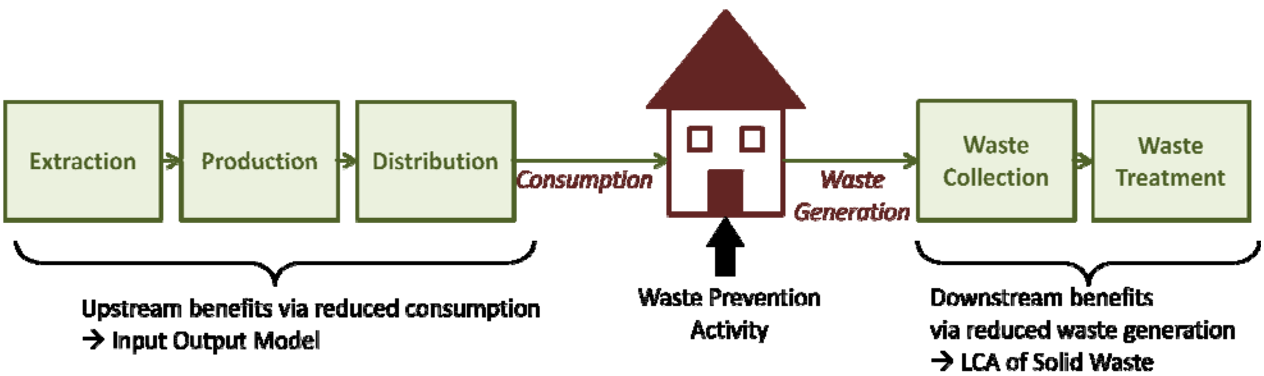


Figure 1: aggregated flowchart of the system under study

In this report the entire consumption pattern of an average household in Trondheim and its related production chains was considered. As a result of consumption a certain waste vector is also generated and enters the waste management system. Figure 1 shows that a Waste Prevention Activity (WPA) at the household level can influence both upstream activities (the production chain) and downstream activities (the waste management system). In this paper, only WPAs that lead to both reduced waste generation and reduced consumption were considered, thus excluding forms of waste diversion such as backyard composting. Indeed, composting decreases waste generation in a sense that it prevents some organic waste from entering the waste management system but it does not affect food consumption in any way.

By definition, a WPA at the household level will affect household waste generation. As a result, consumption categories that do not generate household waste cannot be affected by such a WPA and are therefore not considered in this paper. Categories such as “passenger transport by air” or “electricity” are not material goods owned by households. They are still included in the total consumption pattern but they are not targeted in any of the case studies. As a consequence of focusing on the material side of consumption, the WPAs will affect only indirect emissions: the ones that are not generated directly by households but that occur in the production chain or in the WMS.

In this report several kinds of WPAs are analysed, involving different waste streams and consumption categories. They are analysed thanks to a hybrid LCA model that estimates the environmental benefits generated by WPAs. The consumption side is analysed via an Input-Output based dataset and the waste side is analysed via waste LCA software. The final goals of this analysis are to:

- Compare quantitatively upstream and downstream effects of a WPA,
- Compare the effects of WPAs involving different consumption categories and waste streams,
- Compare the influence of an income effect on each WPA and compare different kinds of income effects.

The term income effect will be defined in details later and refers to the way money saved because of a change in consumption is then re-spent on certain consumption categories.

Given the current uncertainty of combining such different datasets, we only intend to give an order of magnitude rather than precise numbers. This will fortunately be enough to make these comparisons.

1.4. Literature review

As seen earlier, the analysis is split into two sides, the waste generation and the consumption. In reality, there are very few existing attempt that would connect them and analyse them together. In this literature review we will present the authors that have been working on one of these two sides and relevant lessons from their work.

1.4.1. Input-output and sustainable consumption

Many authors have already applied Input-Output models to estimate the environmental impact of household consumption. These studies are in general investigating the entire consumption pattern, but they could as well analyse a smaller consumption basket. Most of the time, they estimate the energy intensity of consumption and the embodied CO_2 emissions.

- **The Dutch Energy Analysis Program (EAP) and energy intensities**

A significant group of authors have been using the Dutch Energy Analysis Program which made a database of the energy intensities and carbon intensities of about 300 goods and services, based on a hybrid framework involving LCA and IOA (Moll et al., 2005, figure 2). The *energy intensity* is defined by the total energy requirements per monetary unit (kWh/NOK, MJ/Euro…). The *emission intensity*, or the *carbon intensity* if we focus on global warming, is the total emissions generated by a products value chain per monetary unit (kg-CO_2/NOK, kg-NOx/NOK…). The emission intensity depends therefore on the total emissions generated by one product, but also on its price. Most of the studies involving the EAP take place in the Netherlands, but Alfredsson (2004) and Carlsson-Kanyama et al. (2005) estimated that energy intensities were similar in the Netherlands and in Sweden and therefore applied the Dutch database on Swedish household. However, there might be some problems when trying to adapt the Dutch database to other countries. The energy mix is very different in Norway for example, whereas there may be more similarities between Netherlands and Sweden.

The Dutch database has the advantage of including a lot of product categories. It is able to give the energy intensities of shoes repairing and to make the difference between tailor-made clothes and ready-made clothes, greenhouse vegetables and root vegetables, buying and renting furniture etc. This allows making high precision scenarios. Besides, Energy intensity appears to be a relevant parameter. For instance eating out can seem to be a non-desirable option compared to a meal at home, regarding its energy use. However, eating out is generally much more expensive and therefore the energy intensity which relies on prices is smaller than a meal at home (8.2 MJ/Euro compared to pork 11MJ/Euro, chicken 14MJ/Euro or greenhouse vegetables 28MJ/Euro) (Carlsson-Kanyama et al., 2005). If we believe that money earned by a household has to be spent somehow, then money spent on eating out will not be spent on some more energy intensive category. Hence eating out can be an option to reduce energy requirements on a constant income.

Alfredsson (2004) investigates the environmental benefits of a “green” consumption pattern on a sample of 1104 Swedish households. She elaborates scenarios based on three consumption categories whose importance in term of CO_2 emissions was highlighted in the literature: food, travel and housing. Reduction numbers were chosen according to the “2021” study by the Swedish Environmental Protection Agency. The green food scenario is based on reduced meat consumption as well as sweets, snacks, soft drinks and alcoholic drinks, and consistent with dietary requirements. The green travel scenario consists in reduction of car ownership, increased car sharing, use of public transportation, walking and biking for short distances, as well as eco-driving. The green housing focuses on reducing water and electricity consumption. Then the three new consumption patterns were also combined into one scenario. The analysis looked at the influence of a marginal income effect (based on the marginal propensity to spend, see 2.2.3) on the environmental benefits.

Scenarios are also extended to 2020 and 2050, taking into account possible improvements in technologies and a growth in income.

The results are not very optimistic since Alfredsson concludes that a greener consumption is not enough to mitigate climate change in a significant way. Indeed first degree analysis (2010 scenarios without income effect) show some environmental benefits that are mostly offsets by the income effect and then by the growth effect. Improvements in technologies (energy efficient houses and cars…) were not enough to achieve great environmental benefits: “If the overall green consumption scenario is adopted and income grows by 1% per year”, it results in a “5% increase in energy requirements and a 7% decrease in CO2 emissions by 2020”.

In their article, **Carlsson-Kanyama et al. (2005)** focus on the direct and indirect energy use of households in Stockholm inner city. Four different kinds of household are investigated with different income level, number of members, surface area and car ownership. In the study, enabling factors, characteristic of the area, are considered: the local support system includes a certain number of retailers with organic food supply, shops with second-hand goods, repair shops and vegetarian restaurants. Options for reducing the energy intensity of consumption were tested (theoretically) on all households. These options involve a change in diet (prioritizing vegetarianism, organic and local food) and a reduced consumption of new clothes, new furniture, trips abroad, car use and newspaper, prioritizing renting, repairing, local culture and taxis (less energy intensive than cars because more expensive).

The result is that combining all of these options while maintaining the expenditure levels of all kinds of household, the reduction potentials stand between 10 and 20% of the total energy use. In particular, options concerning diet can lower food indirect energy use by up to 30%. These results are much more optimistic than Alfredsson’s results, probably because here the income effect is controlled and embodied in the notion of energy intensity. By making sure the energy intensity is reduced and not only the total energy requirements, one includes the influence of prices: the more expensive things we consumes, the less we consume in terms of quantities.

The article from **Moll et al. (2005)** is very complete: it looks at different household types, as for Carlsson-Kanyama et al. (2005), but average households from four different countries are also compared in terms of their expenditures and energy requirements: the Netherlands, the United Kingdom, Sweden and Norway. In order to do that the EAP database was adapted to differences from one country to another, but the adaptation was not complete and this might lead to significant uncertainties.

The results show a high similarity regarding which categories are to be highlighted: “In all countries and for all household types, heating, electricity, food, transport, and recreation are the most important categories with regard to energy requirements”. The results also show differences between household types. Wealthier household have bigger energy requirements on the transportation whereas the highest share of energy requirements for poorer households is the house heating and electricity. Differences arise from the area where households live: being in a city facilitates the access to commodities and thus decreases transportation distances… The city infrastructure therefore matters too.

In this article, the authors point out the importance of studying the household as part of a country. Three determining factors of the differences in household energy requirements are the national electricity generation, the income level of the household and the indirect energy intensities, which are linked to the energy efficiency of the production technologies. The income level appeared to be the most important factor in this study.

- **Direct and indirect impacts of consumption**

Many articles distinguish direct and indirect impacts. “Studies across a variety of countries show roughly equal contributions to energy use from direct and indirect impacts” (Peters and Hertwich, 2006). The same conclusion can be made out of the short literature review in Carlsson-Kanyama et al. (2005) paper. The countries considered there were Norway, the Netherlands, Australia, France, West Germany, Brasil, India, and 11 other European countries in a broader study by Reinders et al. (2003) (it involved again the use of the EAP database). The evidence is thus overwhelming that for households to realize their full energy-saving potential, indirect energy has to be considered.

As seen earlier, indirect energy requirements were highlighted by Moll et al. (2005) and Carlsson-Kanyama et al. (2005). Moll et al. were concerned that taking into account foreign technologies involved in imported goods might result in important errors when estimating indirect emissions. Carlsson-Kanyama et al. (2005) clearly specifies that “future work should include foreign energy intensities when modeling imported goods; otherwise, results may be less reliable”. Peters and Hertwich (2006) point out that issue and stresses it as highly critical: “many previous studies have unrealistically assumed that imports are produced using domestic production technology”.

Modelling imports properly seems to be all the more important in Norway which is a country that imports a lot and that at the same time has a clean energy mix thanks to hydropower. Assuming imported good were made with the same technology as the Norwegian one would tend to underestimate indirect impacts. In their article, **Peters and Hertwich (2005)** used the Chinese technology to model imports from developing countries.

Peters and Hertwich used national and global statistics to build a Multi-Regional framework that model trades between Norway and the rest of the world. Seven major importing partners were selected and the imports from the minor importing countries were then allocated to one of them, assuming similarity between technologies (like China for developing countries). Various sources were then used to map international monetary flows and related emissions of pollutants. Three pollutants were included in the study: CO_2, SO_2 and NOx.

In the end, the impact of imports was compared to the impact it would have if the Norwegian technology was assumed for all imports. The result is stricking: CO_2 emissions are almost three times higher, NOx emissions are 1.5 times higher and SO_2 emissions are ten times higher. The study also compares direct (domestic) and indirect (domestic and imported) emissions. Indirect CO_2 emissions are five times higher than direct emissions, ten times higher for NOx and 70 times higher for SO_2. For these three pollutants, emissions due to household consumption are thus clearly dominated by indirect emissions. The study was also able to highlight important sectors: food, clothing, but also business, transport, refined petroleum, chemicals etc.

The GreenHouse project (Nonhebel and Moll, 2001) tested the influence of a large number of options on Dutch household comsumption and related direct and indirect GHG emissions. For that an IO framework was used, disaggregating consumption into 246 categories and therefore allowing highly detailed scenarios. Options were varied and concerned every aspects of consumption, from mobility to house lighting via changes in diet and clothing habits. A list of 33 main options was made, all succeeding in reducing individually Dutch national GHG emissions by more than 0.5%. Amongst them some were aiming at direct emissions (holiday nearby, shopping on bicycle, wash dishes by hand,) and others indirect emissions (change from synthetic clothes to cotton, more vegetarian diet, sharing daily and weekly papers with neighbours etc.). These options were designed in such way that impact on household behaviour was estimated to be relativity small.

Implementation of all these options was estimated to result in a 27% reduction of the national emissions by the Netherlands. But that was without counting the acceptance by households. A survey was held on 350 households and revealed that the acceptance level would be very low. For a particular option, the maximum level of acceptance was 30%. Less popular options were also the one involving the greatest changes in behaviours. Then the reduction percentage would be 5% rather than 27%. Besides it looks like the income effect (see point 2.2.3) was not taken considered.

The report looked at the reasons that would explain such result. Lack of information is one of them, primarily with regard to indirect energy embodied in products. The report concludes that other sectors of the society have to make changes too and that would allow improving households participation by making accessible and available greener consumption options. “As long as environmental norms are of limited importance within in society as a whole, individual households have not enough opportunities for an environmental friendly behaviour and reduction potentials will never be reached”.

- **Projects**

The articles presented above are all theoretical studies. But real life projects also exist on the topic of household consumption. The **car-free housing project in Vienna** (Hertwich et al., 2008) is one of them. Other car free projects exist and are mentioned in this paper.

In the district of Floridsdorf in Vienna, 244 new apartments were built since the beginning of construction in 1998. As one central condition, tenants were committed to not use or own a car. Instead of building parking spaces for each household, only 20 parking facilities were constructed to be used as car-sharing and bicycle space. Other shared facilities were also built, such as workshops, laundry room, activity rooms, and playgrounds. This project aims at providing an appropriate infrastructure to support sustainable consumption.

In the study from Hertwich et al. (2008), households from the car-free area are compared to a reference settlement which was part of a nearby building complex, with similar characteristics, but without the car-free requirements. The study shows that the total CO_2 emissions of the car-free households were only slightly lower than the reference households. But the carbon intensity however was 20% less: the car-free households had a higher income and they spent it in an overall greener way than the reference households.

Money saved from not owning a car and not paying for gasoline was therefore spent mostly on less emission intensive categories than cars. Residents in the car-free settlements might be more environmentally aware than the average Austrian and local infrastructure was made not only to promote greener transportation but also more sustainable consumption of other categories (food, housing…).

The Perspective project is a Dutch project that performed a practical study on the possibility of reducing energy consumption through “information-induced behavioural change of consumption patterns” (Nonhebel and Moll, 2001). Twelve households participated in a two year field experiment. Their target was to reduce direct and indirect energy consumption by 40%, while obtaining a 20% rise in income at the same time. Households had to fill in weekly lists concerning their energy use and shopping practice and received direct feedback through informative software on the energy consequences of their behaviour. Additionally, advises from tutors were also available to the households.

In the end households almost succeeded in meeting the 40% reduction target. The largest part of the reduction was related to the indirect energy requirements. Important changes observed in consumption behaviour were the extension of products lifetime by repair, changes in diet (less meat, organic food, seasonal vegetables…), changes in the means of transport and travel reduction and the increased purchase of high priced labour-intensive goods (handmade goods…), of high quality and durable goods and of services.

Generally, resulting lifestyle changes were valued positively by the households. The purchase of high quality goods and personal services was much appreciated. The participants were initially burdened with constantly required attention to energy during shopping, however, became accustomed to this aspect. However, the restriction of mobility was never really accepted. Half a year later, a follow-up survey was held: not surprisingly positively perceived changes were kept whereas the others were mostly abandoned. Besides the 20% increase in income was stopped after the experiment and that also explains why the new lifestyle was not fully sustained.

The Perspective project demonstrated the applicability (supported by information and weekly feedback) and acceptability (with an incentive structure) of a not energy intensive consumption pattern. Good news was that a significant part of the behavioural change appeared to be permanent.

- **Other studies**

Many other studies were held on the topic of sustainable consumption using Input-Output frameworks. Tukker et al. (2010) and Reinders et al. (2003) came to the same conclusion as Carlsson-Kanyama et al. (2005) and Moll et al. (2005) regarding the importance of food, housing and mobility as most energy intensive categories.

Other factors influencing consumption were also pointed out, such as time use, with Jalas (2002). The time we spend on one activity and the total amount of time we have available affects our consumption pattern. A rebound effect also occurs if an activity (such as transportation) becomes more time efficient, leaving more time to do something else and consume more of something else.

Duchin (2005) focuses her study on one of the most energy intensive categories: food. The articles reviewed in this study demonstrated that eating less meat is beneficial for both health and the environment (regarding land use, CO_2 emissions and other impacts). The diet recommended by the studies was close to what one calls a Mediterranean diet, characterized by a larger intake of vegetable and a smaller intake of meat and high-fat food than current diets. In order to assess the consequences of changes in diets, Faye Duchin proposes to use her own IO model, the World Trade Model. The input-output matrix, which models the inter-industry monetary flows (the A matrix, see 2.2.2) was extended to include greater detail on agriculture and food production coming from life-cycle inventories.

1.4.2. Potential of waste prevention

In the literature one can find various ways to apprehend waste prevention. This is not surprising when we are aware of all the different aspects of waste prevention activities. Salhofer et al., 2008, gives a good overview. WPAs affect different waste streams and several different groups can be targeted, such as households, retailers, the industry or again public organisms. There are various types of instruments that can be used to influence the target group, such as communication (as for instance awareness campaigns) or making new services or infrastructures available (such as a second-hand store). These instruments can also be economical (taxes on the behaviour that is to be prevented or subsidies on the behaviour that is to be encouraged…), regulatory (implementing laws or product standards) or come from a collaborative agreement (such as certifications and labels).

- **The measure perspective**

Some authors chose to look at the source of WPAs: the measure and the instrument used behind the activity itself. They discuss which measure could be the most effective and why. They study how the measure manages to influence the target group and estimates the resulting participation rate and amount of waste prevented.

This is what **Salhofer et al. (2008)** did. Five case studies are presented and some of them have been conducted in Vienna. They affect five different waste streams: advertising paper, beverage packaging, diapers, food waste and waste from special events. Different measures were tested on the households to prevent this waste. Trondheim already has a reusable plastic bottle system, waste from events is not household waste and diapers are too specific to be studied using high aggregation level data (as it is the case in this project). Advertising and food waste on the other hand are significant waste stream coming from households. Food was already mentioned in the IO part of the literature review as an important category.

In order to calculate the prevention potential, meaning the potential reduction of waste (in kg/cap/yr) through a specific measure, Salhofer et al. used a survey from 140 households. Given the waste composition in Vienna it was estimated that an individual household could prevent a total of 16.5 kg/cap/yr if it refused unaddressed advertising thanks to a "no thanks" sticker. Some households already started to do that and prevent approximately 3 kg/cap/yr, but 13.5 kg/cap/yr are still left to prevent. Two measures were studied: "advertising on request" (household do not receive advertising unless they specifically ask for it) and "information about advertising" which is a campaign that informs households on the ways to cancel and refuse advertising distribution. Given the answers of the survey, it was estimated that the first measure would prevent 5.7 kg/cap/yr and the second measure would prevent 3.3 kg/cap/yr.

The issue of food waste was approached on both the production and retailer side and the consumer side. At the consumer level it was estimated that up to 60% by weight of household waste originates from food, food packaging, food preparation residues and leftovers. Wasted food (food in its original condition, as for example an unopened yogurt, and into only partially consumed food) amounted to 12% (an average of 35.6 kg/cap/yr). Wasted food could theoretically be 100% prevented, since this food was intended to be eaten, in practice though no measure is able to reach a complete prevention. The lifestyle and personal habits of the population have an important impact, as for example the time spent at home or the frequency of meal preparation. Use of a shopping list, providing information on the appropriate storage and handling of food or providing a cookbook with recipes from leftovers are all measures that could decrease this amount. Salhofer et al. do not suggest a number for the prevention potential of food waste.

The amounts of preventable food waste were also estimated by **Quested and Johnson (2009)** working for WRAP (Waste and Resources Action Programme), a non-profit company funded the UK governments. In the UK, almost 65% of the food and drink waste could be avoided, and about 12 extra percents are "possibly avoidable" (classified under that category are bread crusts and potato skin, which can be eaten by some people and not by others). Amongst them, most of the time the food was not eaten on time ("wasted food") or cooked in too much quantity (leftovers). Most drinks end up in the sewer system and not in the WMS, but percentages of avoidable food from the WMS alone were also estimated: 60% of avoidable food waste and 18% of possible avoidable waste. The amounts of packaging involved were unfortunately not considered in this report.

The UK Department for Environment, Food and Rural Affairs (Defra) is involved in WRAP as well as the Waste and Resources Evidence Programme (WREP) established in 2003. An extensive work have been produced as part of this programme on waste prevention and consumers behaviours. Food waste prevention is investigated in WRAP's report (Lyndhurst, 2011) but other kinds of waste prevention are also decribed in Cox et al. (2010) and Cox and Lyndhurst (2009). These reports are much focused on the behavioural aspect of waste prevention: what percentage of people considers that they are sometimes acting to prevent waste, what in our society is a barrier to waste prevention, what kind of measure could be developed to unable it and how effective could they be.

Waste prevention is a complex concept and **Cox et al. (2010)** underlines that "data on the incidence of different behaviours are largely inconclusive". The problem is that waste prevention at the household level encompasses many different behaviours which all tend to be private and invisible, and therefore much less likely to participate in developing a social norm. Surveys investigate people's attitude toward waste prevention were related to different contexts (for instance a specific area or group of people) and therefore difficult to compare and bring out a common behaviour. A few nationally representative surveys are available in the UK, conducted mainly by WRAP (only on food, nappies, home composting, junk mail and single-use bags).

The review by Cox et al. (2010) revealed a general hierarchy in the WPAs' popularity. Donating goods to charity is quite popular whereas activities involving changes in consumption habits are less. Surveys held by the WREP showed that 60% of the consumers do at least one of these activities, "some of the time", which is not very conclusive. Through these studies one can however start to understand better what the main behavioural barriers are for waste prevention.

Other papers evaluate the efficiency of one single measure, such as Björklund and Finnveden (2007), who looked at the effect of introducing a tax on incineration and Sharp et al. (2010) who looked at the potential of intervention campaigns. This topic will not be developed more since the measure and the behavioural reason behind a WPA were not at the center of the present analysis; though some data from Quester and Johnson (2009) and Salhofer et al. (2008) were used to evaluate the realism of the scenarios.

- **The environmental impact assessment perspective**

Some other authors have chosen to examine the environmental benefits of a waste prevention activity, regardless of the measure that generated it. They assumed that a certain amount of waste was prevented somehow and they looked at the resulting environmental impacts. According to Cleary (2010) and Gentil et al. (2011) the measure perspective was studied much more often than the impact assessment perspective: "Waste prevention has been addressed in the literature in terms of the social and behavioural aspects, but very little quantitative assessment exists of the environmental benefits" (Gentil et al. 2011).

There are different methodologies that were developed in order to deal with waste prevention. They were discussed and compared in my project. In his paper, Cleary (2010) evaluates the methodological issues of assessing waste prevention with LCA tools and propose a methodology that would get around these issues. Ekvall and Weidema (2004) are supporters of the Consequential LCA methodology and present it in their paper.

Gentil et al. (2011) uses the methodology recommended by Cleary (2010) to assess the potential of partial prevention of unsolicited mail, beverage packaging and food waste. This potential was evaluated on both a "high tech" WMS and a "low tech" one, that is to say one that has a high recycling rate and one that has a high percentage of landfilling. He assumed realistic prevention potentials for the three waste streams partly using Salhofer et al. work (2008): 20% for both food waste and unsolicited mail and 60% for beverage packaging. Gentil's framework incorporated avoided energy production and avoided material production to the waste management system. Waste prevention was seen as an equivalent to waste treatment that has no environmental burden (no direct emissions in the waste prevention process) but indirect environmental benefits from avoided production. The same way incineration with energy recovery is decreasing the need for other sources of energy, waste prevention is decreasing the need for new products. Production stands here downstream of the WMS (it does not change anything compared to our model *a priori*, only the way the system is represented).

The WMS studied are the ones of a fictitious city that produces 100000 tons of waste yearly. The combined prevention measures reduced it by 1180 kg paper, 950 kg plastic, 4570 kg glass, 4900 kg vegetable waste and 1540 kg meat waste. For each WMS two sets of results are compared to a baseline scenario where no WPA is undertaken. The first set of result does not take into account the impacts avoided at the production level, the second does.

The results show not surprisingly that the high tech WMS generates much less environmental damage than the low tech for most impact categories (it even generates environmental benefits: numbers are negative), except for human toxicity water. The numbers for this impact category are similar for all first sets of results and this is explained by the extra need of cleaning and transport compared to the low tech WMS. The results for waste prevention excluding avoided production are slightly higher for this impact because of the need for detergents for cleaning the reusable bottles. But this small increase is compensated when the benefits of avoided production are taken into account. Results also show that waste prevention does not offer a lot of environmental benefits compared to the baseline scenario when avoided production is not included. When it is, waste prevention has much more environmental benefits and five out of eight impacts are greatly reduced (global warming, acidification, ecotoxicity water chronic, nutrient enrichment and human toxicity soil).

Olofsson carried out in **2004** a consequential LCA where waste prevention is compared to recycling regarding global warming potential. The waste prevented or recycled is municipal solid waste and is composed of paper and cardboard (from newsprint, office paper and corrugated cardboard), glass and metal packaging. The system studied is representative of Sweden and forecasts the period between 2008 and 2012. Three scenarios are compared. The baseline scenario illustrates current development: a large increase of incineration and a smaller increase of biological treatment compared to 2004. In the waste prevention scenario, waste is diverted from incineration and landfilling. In the recycling scenario, the same amount (and composition) prevented is instead recycled. The amount of waste prevented is an arbitrary (yet realist) assumption; Olofsson is therefore not interested in the WPA itself but only in its result.

Consequencial LCA provides an extended framework that incorporates any relevant process that is related to the main system and that the practitioner wants to study. The system here includes the energy generation processes because they are affected by a change in the amount of waste incinerated via energy recovery. The “rest of the world” WMS is included because it is linked to the Swedish WMS via the international scrap market with a limited demand for recycled material. The marginal waste management in the rest of the world is approximated to landfill disposal since this is the most common waste treatment. The Swedish energy technology was assumed to be natural gas because it represents an average between highly polluting energies like coal and renewable energies like wind power. The marginal energy mix in Sweden was thus assumed to be equivalent to a 100% energy production by natural gas. Marginal processes were used because they are more appropriate to model change.

The consequences of waste prevention on the system are divers. The most important one is that it leads to a reduction of both virgin and recycled production, because of reduced material consumption. This leads to the most reduction in greenhouse gas emissions. Waste prevention also leads to reduction of waste treatment and therefore reduction of emissions related to it. It let the recycling constant in Sweden and thus does not affect the scrap market and the rest of the world. The only drawback is that incineration with energy recovery in Sweden is decreased and this energy has to be substituted by natural gas, leading to more greenhouse gas emissions in the energy system. In total, for a prevention of 4% of the Swedish municipal solid waste, the emissions of GHG are reduced by 5 to 9% of the Swedish national reduction goal (this corresponds to a reduction of 150 to 270 kilotonnes CO_2-eq). Recycling is less beneficial, especially as it reduces the demand of recycled material from the rest of the world and thus increased landfilling there.

Gentil (2011) and Olofsson (2004) therefore both conclude that the main benefits of waste prevention come from avoided production.

- **Other studies**

Farrant et al. (2010) made a product LCA to assess the environmental benefits of reusing clothes. As in every product LCA, single life cycle chains were studied (not entire consumption categories as in IOA): the one of a cotton T-shirt and the one of 65% polyester/35% cotton trousers. The analysis compares the environmental impact of reusing these two garments with the environmental impact of incinerating them and buying new ones. It is considered that there could be a significant environmental burden of reusing clothes in the case where they are transported to be reused in developing countries.

The approach is interesting in the sense that it investigates the behavioural relationship between buying second hand clothes and buying new ones. Indeed it is not straightforward that buying 1 second-hand garment will automatically prevent the consumer from buying 1 new garment. First second hand clothes are cheaper and purchasing them might generate money savings that could potentially still be spent on new clothes. Besides they are obviously not perceived the same way by the consumer: they are not interchangeable. Farrant et al. conducted surveys targeting second-hand clothes consumers in order to get information of this particular relationship.

This study therefore considers both the behavioural aspect and the environmental impact aspect of waste prevention. It also considers both the production side impacts and the waste management side impacts. However this is only done for two products. This results in some relevant effects that are not considered, as for example the income effect, which is visible only when considering the entire consumption pattern.

The surveys revealed that the purchase of 100 second-hand garments could prevent from buying between 60 and 85 new garments. Besides, it was assumed that among 100 collected garments 60 would be reused, 30 recycled, and 10 would be discarded, which was estimated by the Humana People charity in both Sweden and Estonia. These assumptions were used in the LCA scenarios. The LCA results finally showed that the collection, processing, and transport of second-hand clothing still had negligible environment impacts and furthermore generated significant savings by replacing virgin clothing: the collection of 100 garments for reuse generates a 14% decrease of global warming for the cotton T-shirt. Human toxicity is also decreased by 45% when reusing the polyester/cotton trousers.

A number of studies that are all concerned with estimating the environmental damage of household consumption have been reviewed. Many different approaches were chosen (downstream of upstream approach, focusing on global warming potential, energy use…), using different methods (IOA, product LCA or waste LCA, surveys…) and having different detail levels (an entire consumption pattern, a consumption category, a waste stream, one single product…). They all seem to point out significant potential environmental benefits from a change in consumption behaviours even though some of them are more optimistic than others: the need for help from the industry or the government to unable and facilitate these behaviours is often mentioned.

All of these studies are relevant for our topic and they all show various aspects that are important to consider when assessing waste prevention and sustainable consumption: income effect, growth effect, technological improvement, infrastructure availability, behavioural barriers... The assumptions taken as well as the results from these studies should be kept in mind for the rest of our study.

2. The Hybrid LCA framework

2.1. The framework

2.1.1. Hybrid LCA and Waste Management LCA

The framework applied in this Master thesis is a hybrid LCA framework which combines an Input-Output Analysis and a waste management LCA. These three methods will be first defined quickly.

- **Input Output Analysis (IOA)**

Input–Output Analysis is an economic modeling technique that describes monetary transactions in a defined system (a group of industries, a country, the world…) and reflects the complex interdependencies between industries. The data used is derived from “make” and “use” matrices reported in national statistical bureaus. These official Input-Output Tables (IOTs) cover the entire economy using detailed government statistics. In order to describe the embodied environmental impacts of national or international transactions, IO frameworks must be extended to incorporate environmental data and connect them to monetary flows.

Here IOA is used to analyze the consumption side of our system. IOA is a very appropriate tool to study household consumption; first because numbers are expressed in monetary units, therefore a significant unit for the consumer; and second because IOTs provide the right level of details. Indeed, classic LCAs typically look at the life cycle impact of one product whereas IOAs have a higher aggregation level which allows us investigating many product categories at the same time. The level of detail in classic LCAs is too high to study an entire consumption pattern.

One must however point out that IOA also has some disadvantages in our study. The environmental extension is unfortunately not as complete as the one of a classical LCA, meaning that the environmental stressors reported are fewer, and the aggregation level prevents us from building more precise scenarios. These limitations will be developed at the end of this chapter.

- **Waste management LCA**

This kind of LCA is meant to study the environmental impact of a Waste Management System (WMS). A WMS is defined by different means of waste collection and different sorting, treatment and disposal methods, as will be discussed later. A waste management LCA studies the environmental impact of a certain quantity and composition of waste entering a certain WMS.

The WMS used in our analysis is the WMS from Trondheim. It included local facilities such as the paper sorting facility and the incineration plant, as well as transportation to facilities in other regions and other countries. It incorporates every process involved in the management of Trondheim’s household waste. Our framework also incorporates future use of the waste: use of recycled material, electricity and heat, therefore extending the system beyond the WMS and including secondary production.

- **Hybrid LCA**

For the purpose of this study, the IO dataset and the waste management dataset were incorporated in what one may call a hybrid LCA. This method uses an LCA foreground, with physical flows and detailed individual processes, linked to an IO background. The LCA foreground here represents Trondheim's WMS. Using the words "foreground" and "background" might be misleading here because it somewhat implies different levels of importance. Here however, the IOA and the Waste Management LCA are both as important. The foreground system has however a higher resolution, which makes sense since it represents a city's WMS whereas the IO dataset represents global trade flows.

The advantage of hybrid LCA is to combine the benefits of both IOA and LCA by covering the entire economic boundaries while maintaining detailed process based life cycle inventories on the selected part of the system (here the WMS). It has the benefits of using the precision of an LCA data set but embedding it in a broader economical system provided by IO data (Suh, 2004). Some inputs are often missed in process based LCAs such as services but also products such as office stationary, computers, and furniture, and also more indirectly used products such as food, textiles, footwear, soap and detergents etc. IOA describes all of these connections as LCA compensates for its lack of detail in the selected foreground. More details on the advantages of hybrid LCA in this study were presented in a previous project (Lèbre, 2011).

- **Double counting**

For a hybrid LCA framework to be valid, one must first make some adjustments to avoid double counting. Indeed it is not enough to combine both datasets together. While the Input Output dataset describes the entire monetary flows in and out of the Norwegian economy, the Waste LCA dataset describes a part of it that is already taken into account in the IO dataset. One must therefore remove the waste management from the IO dataset, or else it will be counted twice.

In our case, avoiding double-counting is simply done by removing the waste management expenses out of the household expenditures. In the Consumer Expenditure Survey, the "refuse collection" category is then set to 0 Norwegian kroner.

2.1.2. Framework description

The following figure shows the different parts of the framework and their connections via a simple example of man clothes:

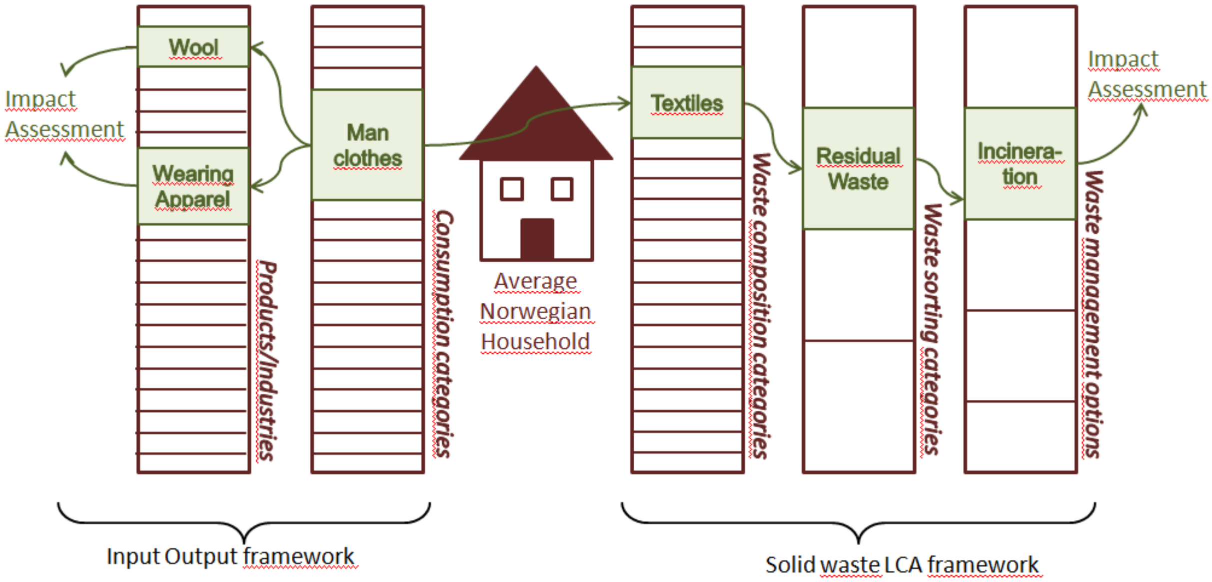


Figure 2: the framework, correspondence between datasets

The consumption occurs at the household level when the household purchases a certain amount of goods or services, here “man clothes”. The consumption category refers to other products in the IO tables, “wool” and “wearing apparel”, and from there the impact assessment can take place, evaluating the consumption side of the framework. On the other side, waste is generated as “textiles”, along with other clothes and house textiles. It is not presorted at the household level and is collected with the residual waste and transported to the incineration plant where energy is recovered and supplies mostly the Scandinavian NORDEL grid. The Waste LCA assesses this part of the system from waste generation to energy production in the incineration plant.

In this framework separating the supply chain and the disposal phase, one may wonder where the use phase is. The use phase is responsible for significant direct environmental impacts. But these emissions are actually taken into account in the household’s expenses. Indeed, direct emissions in the use phase are mostly due to fuel consumption for mobility and heating (which is, in Norway, most of the time provided by electricity which has no direct emissions). Purchases of fuel and electricity are included in total household consumption, and therefore counted in the total emissions. Electricity purchase does not lead to direct emissions but indirect ones occuring upstream in the supply chain. The framework as it is now does not allow us to compare direct and indirect emissions as in some of the studies quoted in the literature review, but it is still possible to extract fuel consumption and other possible sources of direct emissions from the total expenditure and assess them separately.

2.1.3. The data

Several datasets were needed to build the framework. The following figure gives a good overview of the multiple translations that were necessary to link them together.

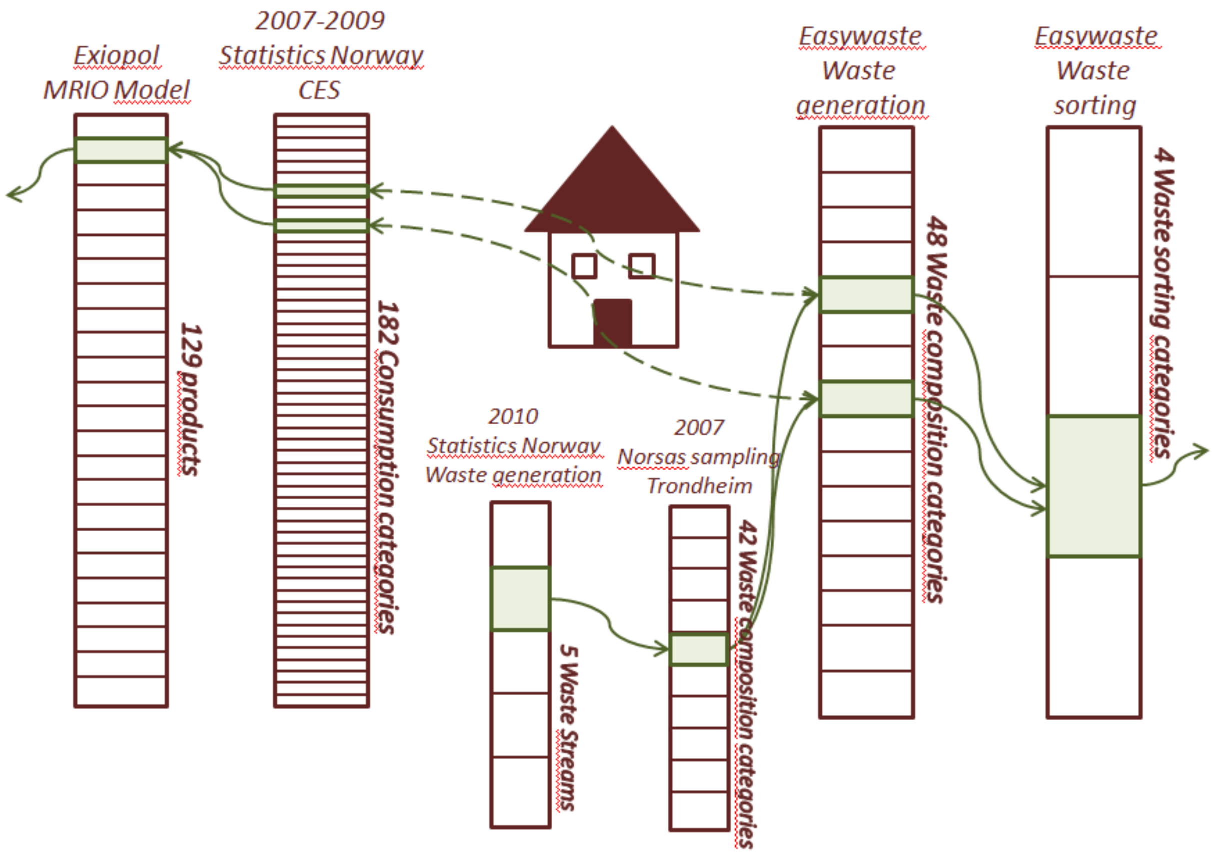


Figure 3: the data, translations from one dataset to another

The two main datasets are the Norwegian Consumer Expenditure Survey (CES) and the Waste Generation Vector (WGV). The latter was built using three different sources: the statistics Norway website, the Norsas waste composition analysis and the Easewaste template. The resulting vector then was introduced in the Easewaste software that sorted it into the desired number of sorting categories, following the sorting efficiencies that were defined. Once the waste management technologies are defined as well the software calculates the resulting environmental impacts.

On the consumption side, the CES must be translated to fit the MRIO model, so that the environmental analysis can be made. More details are explained in the following paragraphs.

2.1.4. The household

The household is at the centre of the system as its consumption and waste generation are analysed. All numbers concerning the household correspond to an "average" household activity during one year: the total expenses and waste amounts generated in a year. It is however important to specify what is meant by an average household. An average household is characterized by an average consumption and an average waste generation: that is to say the total consumption and the total waste generation, divided by the number of inhabitants in Trondheim or in Norway depending on the data source. The number of people in this household is again an average number: number of inhabitants in Trondheim divided by the number of households in Trondheim, and this ratio is equal to 2.2. But this is not enough and the underlying assumptions of studying such household should be pointed out.

- **Life span of goods**

The household under investigation is not strictly speaking a “real” household. First, a real household cannot possibly have 2.2 inhabitants. Besides, a real household would not buy books and throw them away immediately afterwards. Books take usually years to be discarded. The same goes for all the durable or semi-durable goods, like cars, houses, furniture, utensils etc., basically most of the goods purchased except food. If it were a real household, the life span of a good and the year considered for the study would therefore matter a lot: they both affect the content of the CES and the waste generation. Estimations would have to be made, such as: “a book bought in 2008 is expected to be discarded in 2013”.

Fortunately, our household is not “real” and is the sum of all households. And, amongst all of these households, some of them buy new books, some of them throw these books away, others sell them via second-hand stores, and others buy them second-hand. And this is all shown in the average CES and Waste Generation Vector (WGV). A year has to be defined for the study because purchasing power, waste generation and population evolve from one year to another, but the life span of a good is not important anymore. Of course the life span of a good can evolve with time too but that is another matter and it involves studies of consumption and waste generation over a much longer period of time than one year.

This assumption makes a big difference between a classic LCA and our model. A classic LCA might have difficulties in attributing the benefits of a waste prevention activity. For example a household gives away its clothes to a charity, and another household gets the second-hand clothes. A classic LCA would tend to attribute the environmental benefits of reduced consumption of new clothes onto the second household, even if the first household did the initial action of diverting clothes from the waste system. In our model this attribution issue is not relevant, since the average household incorporates both households.

One can therefore reasonably assume that consumption and waste generation occur simultaneously and are proportional. It allows us to say that there is a direct correspondence between the amounts of good purchased and the amounts discarded. While a household buys a certain good a second household throws away the used version of this same good: on a material flow point of view, CES and WGV are equivalent in terms of weight (see equation below). This was a key assumption to build the scenarios.

$$Waste\ Generation\ (kg) = \ Purchases\ (kg)$$

- **The total expenditures**

The average household is characterised by an average yearly income. As a consequence of the assumption above, this income is entirely spent within a year. In practice, people tend to save a part of their yearly income to spend it on the next years or even much later (their retirement plan for example). In theory though, money will sooner or later be spent. And since the household considered is fictitious, all savings made by one household are spent at the same time by another one and both are included in the CES.

$$Revenues\ (NOK) = \ Purchases(NOK)$$

2.2. The IO side

2.2.1. The Norwegian Consumer Expenditure Survey

Norwegian CES or "Household Budget Survey" is a list of yearly household expenses. They have 182 consumption categories, representing the entire household expenses, from electricity bills to education expenses or again food purchases. These categories are very similar to the ones used by Eurostat. For the current CES, 2200 people between 0 and 79 years old are selected, not including people living in institutions (Andersen, 2001). The household of which a selected person is a member is then asked to participate. These households do not have to record their expenses for an entire year, only for two weeks. During a two-week period, 1 out of 26 of the sampled households is contacted for interviews and asked to record their current expenditures. The first data collection therefore lasts 26 times 2 weeks: one year. This is then repeated the next two years with the same households. The CES published is the average of these three years.

Data is collected the following way: a first interview is held before the start of the account period, mainly for the collection of background data: age, sex and number of people in the household as well as income level. Then, during two weeks, the household records its daily expenses in several account books. Each household member over 15 years old keeps special accounts of its own expenses in one of the account books. A second interview after the account period is used for recording purchases and ownership of durable goods, as for example certain goods related to housing, private transport equipment, some electrical appliances, expensive clothing or again package tours.

The CES used in this paper is the most recent one and is an average of the expenditures in 2007-2009 with 2009 prices. It can be found on the Norwegian statistics website (Statistics Norway, 2012). Data is available for many different types of households (single people, couples, couple with one child etc.), from different regions and from different density areas (distinguishing big cities from country-side).

Unfortunately there was no table available for a CES on Trondheim with the maximum level of detail (182 categories). The maximum level of detail was necessary in order to be able to connect the CES to the Input-Output tables and thus make the calculations (see part 2.2.4 about the "CES to y" matrix). The CES for an average Norwegian household was therefore chosen. It can be found in *appendix 1* along with a CES of Trondheim, Bergen and Oslo with lower detail level.

2.2.2. The Exiopol Multi-Regional Input-Output Model

In order to calculate the environmental impact of the average yearly Norwegian consumption, it was connected to a Multi-Regional Input-Ouput Model built by Exiopol. Exiopol is a project funded by the European Commission gathering researchers from all over Europe. NTNU is involved via the active colaboration of Edgar Hertwich.

In their “top-down” approach (Exiopol, 2010), Exiopol developed this new environmental accounting framework called the EXIOBASE. It is a Multi-Regional Environmentally Extended Input-Output Table, meaning it describes national and international monetary flows between sectors and provides data on the environmental stressors embodied in them (emissions, resource use etc.). It is centred on the European Union but still covers a total of 43 countries in and out of Europe, all of them representing 95% of the global economy. The 44th country represents the aggregated version of the rest of the world, which is more than 150 other countries.

Two sets of matrices are available, one based on industry sector and the other one on products, both having 129 categories (129 products or 129 sectors). The latter one was chosen for this analysis because products have a more meaningful role when dealing with household consumption. The set of matrix is represented in the figure below:

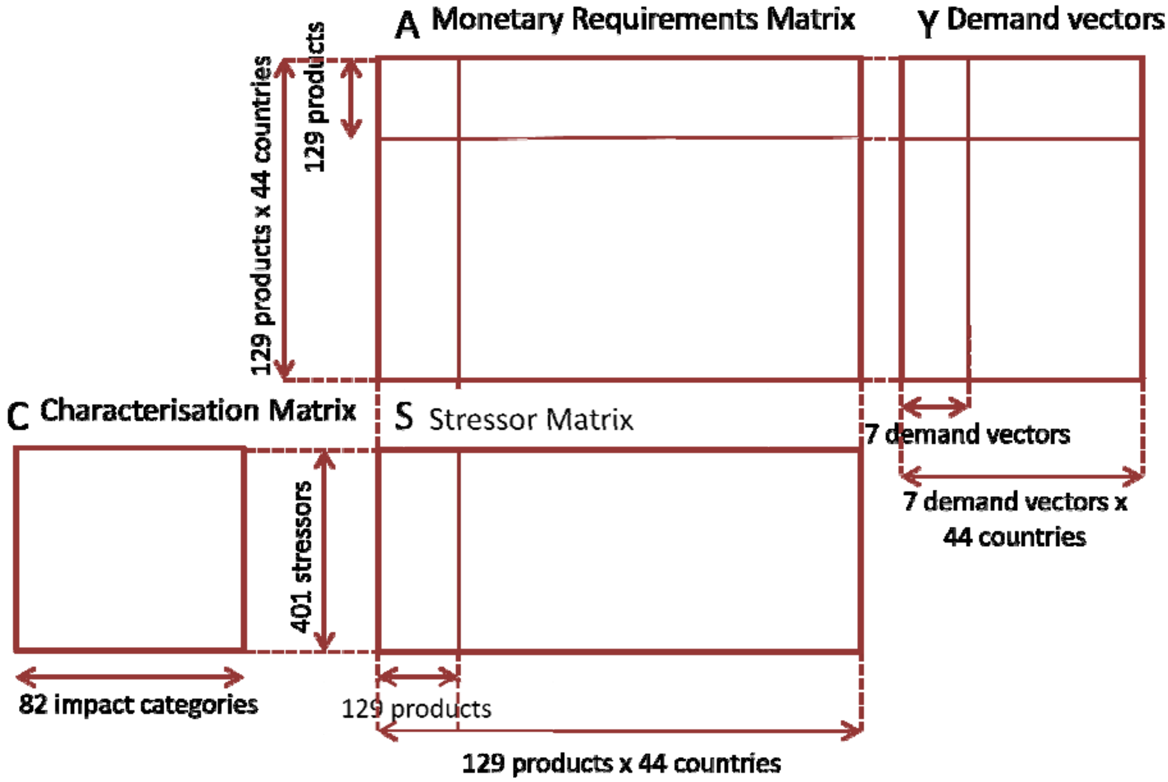


Figure 4: the Input-Output model, main matrices

The A matrix, or the “requirement” matrix represents monetary exchanges between countries and between products. It is derived from the total flow matrix Z which contains the entire monetary flows occurring in a year. Z is then normalized by the global yearly production to obtain A. In A, flows are therefore all expressed in “million Euros per million Euros of production”. The 129x129 square matrices located on the diagonal of A are the domestic monetary exchanges of each country. The rest are international flows. For instance, the 129x129 square matrix positioned on the 41st group of column (out of 44 groups of 129 columns) and the 2nd group of lines represents the imports to Norway from Belgium.

The Y matrix is the juxtaposition of all the demand vectors for each country. The demand vector of a country is decomposed into five main “consumers”: households, non-profit organisations, governments, gross fixed capital formation and exports. Two more columns are for “changes in inventories” and “changes in valuables”.

The Stressor matrix or the “Factor” matrix expresses the generation of stressors per monetary output of products from each country. The numbers are unitary and correspond to flows per million Euros, as for the A matrix. The 401 stressors, it contains are environmental emissions and resource use (water, land etc.), as well as economical and social factors such as value added and employment. The Stressor matrix allows distinguishing technologies and energy mixes of countries. Indeed a country that has an energy mix overpowered by coal or inefficient production technologies will have unitary emissions in the Stressor matrix higher than other countries.

Finally the Characterization matrix gives the correspondence between stressors (for example: kg of CO_2 emissions released in the air per million Euros of production of product # in country #) and environmental impacts (for example global warming potential). Stressors, or “extensions” as they are called in the Exiopol project, are aggregated there to compile indicators such as Global Warming Potential, Acidification, Human Toxicity, Total material requirement, and external costs.

- **The importance of imports**

The EXIOBASE was meant to help answering questions related to the social, economical and environmental impacts of international monetary flows. It is able to assess the external costs of global economic production and to evaluate the impacts of policy interventions in various areas such as buildings, mobility, and food.

Thanks to its distinction of what is produced and consumed domestically and what is imported, the EXIOBASE allows to evaluate the impacts embodied in imports to Europe for example. Which part of what we consume is imported and what is the share of environmental impacts that comes from imports? In the present case, this is a big advantage of using the EXIOBASE in our analysis of Norwegian household consumption. As pointed out earlier, to each country is allocated a stressor matrix which contains environmental stressors per units of its domestic production and this is characteristic of the country’s production technology. Norway being a developed country which has a high share of its energy mix coming from hydropower, its domestic production is relatively clean compared to other countries that use much more fossil energies. In a standard IOA one may choose to use a “mirrored economy” assumption, which assumes that the foreign production technology is the same as the domestic technology. In the case of Norway, this would lead to significant mistakes as it would underestimate the impacts embodied in imports.

2.2.3. Rebound effects

When doing an Input Output analysis, one should be aware of the possibility of having side effects that may influence the results of the analysis. Most of these rebound effects are behavioural and therefore difficult to anticipate. There are various types of rebound effects related to Input Output analysis and here are presented the most relevant ones.

- **Income effect**

The income effect is the rebound effect studied in this paper. It is probably the most related to household consumption. Indeed, consumption highly depends on the income level. A household that earns more money will consume more, of course, but will also consume differently than a poorer household. The income effect reflects how a change in income influences consumption. If a household suddenly earns more money, how will it spend the extra money? And if a household earns less money, on which consumption category (or categories) will it decide to make savings? Money saved or spent is not usually distributed evenly on every consumption categories. Health or education categories are typically categories that are not much influenced by changes in income (as long as the changes are not too extreme), whereas leisure, holiday travelling and clothing can be highly affected. Some categories are more important to a household than others and this affects how the money is spent or saved. One may think it is more important to eat healthy food than to wear nice clothes and will therefore rather make savings on clothing than on food in case of a reduced income.

The income effect therefore depends highly on the consumer's choice. But it also depends on the amounts involved and on the unit price of a category: if one gets 1000NOK extra, one might spend it on clothes of restaurants, whereas if one gets 10 000NOK extra, one might spend it on a holiday trip abroad. It also highly depends highly on what we call the **emission intensity** (or carbon intensity if we focus on global warming) of the consumption categories involved.

In this paper, income level is assumed to be constant and is entirely spent in each scenario. There is therefore no matter here of earning more or less. However, the problem is that our scenarios are about reducing consumption and therefore reducing expenses on a selected consumption category. And according to our assumption the money saved by doing so has to be spent somewhere else. The question is then again where this money will be spent. Answering this question accurately is important because the final results of the analysis will highly depend on which category the savings are spent. If this category is more emission intensive than the one on which money was saved in the first place, then the scenario will not only fail in reducing carbon emissions but will increase them! Spending saved money on holidays abroad, which is a realistic option for wealthy households, is very likely to have this effect, since travels by plane are extremely carbon intensive. Alfredsson (2004) in her paper investigates the effect of greener consumption behaviours. She applies a rebound effect for the money that has been saved in the process. This single rebound was enough to significantly reduce all the environmental benefits, and sometimes even create more environmental damage. She then looks at the effect of a growth in income and the results are even more negative. An increase in income, as well as would a reduction in prices, has the same effect the initial savings since it allows consuming more. In the present model however the prices and the income level were kept constant.

In Hertwich et. al (2008) case study of the car-free settlements, the income effect was also considered. Here the effect was smaller, simply because the car use is a very carbon intensive activity. It was estimated that the rebound effect occurred on three main categories: air transport, nutrition (eating out mainly), and “others”. The category “others” is only 14% as carbon intensive as car transportation, so a rebound on it would not compensate the environmental gains of using less cars. Nutrition and above all air transport are however more carbon intensive. Fortunately only little extra money was spent on air transport; and eating out still has relatively low carbon intensity since it is quite expensive compared to other forms of nutrition. In this project, the income-based rebound effect was estimated to be “on the order of 10-20%”, that is to say that we can observe a 10 to 20% increase of the CO2 emissions compared to a scenario without rebound.

Both of these case studies show how the income level can affect the environmental gains and why it can be necessary to consider it. Aware of that important side effect, the consumer could potentially make sure to spend the savings on some “greener” consumption category and this possibility was included in some rebound scenarios. But the only way this income effect could be completely erased would be to have the household voluntarily reduce its income. Bigger issues would then have to be raised such de-growth and alter-globalization but that is not the scope of this paper.

- **Other effects relevant to Input Output Analysis**

Other rebound effects exist though they were not considered in this paper. Looking at the economical aspect of Input Output Analysis, the **growth effect** is very relevant. It comes from the fact that an increase in technical efficiency also reduces costs to the industry, which leads to a reduced price in goods and services and hence an increase in demand for these goods and services. Increases in efficiency will therefore lead to a lower than expected environmental gain, or even sometimes to an increase in environmental damages because reduced costs lead to increased activity levels.

There are many other kinds of side effects that can happen when it comes to changing people’s behaviours, and it is impossible to list and predict all of them. When encouraging people to adopt a greener behaviour, through awareness activities for example, some unexpected side effects can occur. For instance, people could be tempted (consciously or not) to consume more of a greener product if they are provided one: people may drive their car more often if they switched to a more eco-efficient one. Knowing a product is recyclable may influence them into consuming more of it: people might think “I recycle so I can consume as much as I want”. This is the danger of emphasizing recycling without introducing the concept of waste prevention and sustainable consumption.

But not all rebound effects are necessarily negative. Technological Spill-over can sometimes be beneficial. For example, in the U.S., hydrogen in fuel cell cars is more polluting than diesel-hybrid cars, but in Europe it is the contrary. The reason is that hydrogen compression and transport uses electricity, which is much less carbon intensive in Europe and it is therefore better to use electricity than hybrid fuels (Hertwich and Strømman, 2011). Improvements in some areas (the energy mix of a country) can hence make possible new technologies to develop which would bring further improvements. Here there is a positive side effect of a positive initial action.

Other positive spill-over effects can take place at the consumer level. For example, an environmental awareness campaign focused on the importance of recycling could have unexpected positive side effects: more aware of environmental issues, people would be more inclined to also change other parts of their behaviour: eating less meat, saving electricity at home, using more public transportation. Once a more conscious behaviour is created it can be easier to continue and improve it even more. Another example: consumers are easily influenced by other consumers. Hence if one person begins to buy organic food, his or her relatives may follow the same path.

- **Marginal and simple income effects**

In this paper, for each consumption category investigated, several income effects were tested. The money saved was distributed on one or more other categories. Rebounds on a service, such as restaurant, repair of shoes and clothes, holidays and some cultural activities, were applied. These scenarios will have no incidence on the household waste generation but will influence the upstream environmental impacts.

For each scenario the influence of a more general rebound effect was also tested: the money saved is re-spent on every consumption categories, except obviously the one whose expenses were reduced in the first place. The amounts distributed to each category are proportional to the size of the category, that is to say its share of the total expenses: the more people spend on a category, the more they will spend the money saved on it. The following equation specifies how the distribution was done. $Expenses_j^i$ are the expenses for category i in scenario j. $Expenses_{ref}^i$ are the expenses for category i in the reference scenario (the Norwegian CES with no modification).

$$Expenses_j^i = Expenses_{ref}^i + Money\ saved \times \frac{Expenses_{ref}^i}{Total\ income}$$

This rebound effect was preferred to a simpler rebound effect that would distribute the money saved evenly to every other consumption categories (see the following equation):

$$Expenses_j^i = Expenses_{ref}^i + \frac{Money\ saved}{Number\ of\ categories}$$

This rebound effect is less realistic than the previous one because it makes no difference between categories. It assumes that the household would re-spend the same money on “bread” and “footwear for men” for instance. The first rebound effect reflects better the unit prices of each category: the amounts of money spent on each category are not the same simply because some categories are more expensive than others.

However, there are even more accurate ways to model the income effect. In her PhD thesis Alfredsson (2004) uses the Marginal Propensity to Spend (MPS). The term “marginal” is used to qualify a small and singular change, as opposed to “average”: if one earns one more euro compared to its previous income, how will this euro be spent? The MPS can be calculated using consumer expenditure surveys for various income levels. By comparing two surveys that have different income levels, one notices the differences in consumption patterns. The MPS reflects this difference, as can be seen in the equation below. The "total change in expenditures" is equal to the difference between the two income levels.

$$MPS_i = \frac{Change\ in\ expenditures\ on\ component\ i}{Total\ change\ in\ expenditures}$$

This way of modelling income effect is the most accurate because it not only takes into account current expenditures and unit prices of one category, it also takes into account the consumers preferences: if one has more money, where is one likely to spend it. Looking at CES with different income levels helps determine these preferences. This rebound effect was also tested on all scenarios. Results with this **"marginal" rebound** were then compared to the results with the former rebound presented above. See *Appendix 2* for the calculations of MPSs.

2.2.4. Calculations

The calculations are the same for all scenarios. For each of them the CES vector was first modified, applying first reduced consumption in the selected categories and then a rebound effect so that the total level of expenditure stays the same. Then the CES vector must be translated into a y vector that is 129 lines by 1 column, hence adapted to the IO dataset. In order to do so, a "CES to y" correspondence matrix was provided, which was made by Kjartan Steen-Olsen, PhD candidate in the Industrial Ecology programme at NTNU under the supervision of Edgar Hertwich. The CES vector just has to be multiplied by this matrix to obtain a y vector.

The Norwegian household demand vector was extracted out of all demand vectors from the IO dataset. It is in the first column of the 41st country in the Y matrix, that is to say the 281st column (see figure 4 above). This vector (5676 lines x 1 column), representing households' final demand for commodities in 2010, was used to obtain the proportions of each commodity that comes from each country. A new vector was made (5676 x 1 too) that expresses these proportions as percentages (for example, for a certain demand of processed rice from Norwegian households, 10% comes from Ireland and 58% from the ROW…). This percentage vector was multiplied to the y vector to get final (5676 x 1) demand vector, so that y is distributed on the 44 countries. The multiplication was made possible by duplicating y 44 times and diagonalizing it (see equation below):

$$Y = \begin{bmatrix} y & \cdots & 0 \\ \vdots & \ddots & \vdots \\ 0 & \cdots & y \end{bmatrix} \times \begin{bmatrix} \% \\ \vdots \\ \% \end{bmatrix}$$

Where Y is a 5676x1 vector, which can now be used to calculate its related environmental impacts, as shown in the following equation:

$$d = C' \times S \times (I - A)^{-1} \times Y$$

d being the impact vector, $C'$the characterization matrix transposed, S the stressor matrix and $(I-A)^{-1}$ the Leontief Inverse. I is a 5676 by 5676 identity matrix and A is the requirement matrix, as defined in paragraph 2.2.2.

2.3. The Waste Management side

2.3.1. Easewaste and the Waste Management System

Data on the waste treatment system in Trondheim was provided by Helene Slagstad, who is a PhD candidate at the department of Hydraulic and Environmental Engineering at NTNU, under the supervision of Helge Brattebø. She built the system on Easewaste, LCA software designed for waste management by the Technical University of Denmark. To make the waste scenarios, one has to enter the waste quantity and composition, the sorting efficiencies, define the treatment system using either pre-defined processes or new ones, and then Easewaste makes the impact assessment. For all scenarios the treatment system was the same, only the waste characteristics changed. The following figure shows a summary of the WMS in Trondheim:

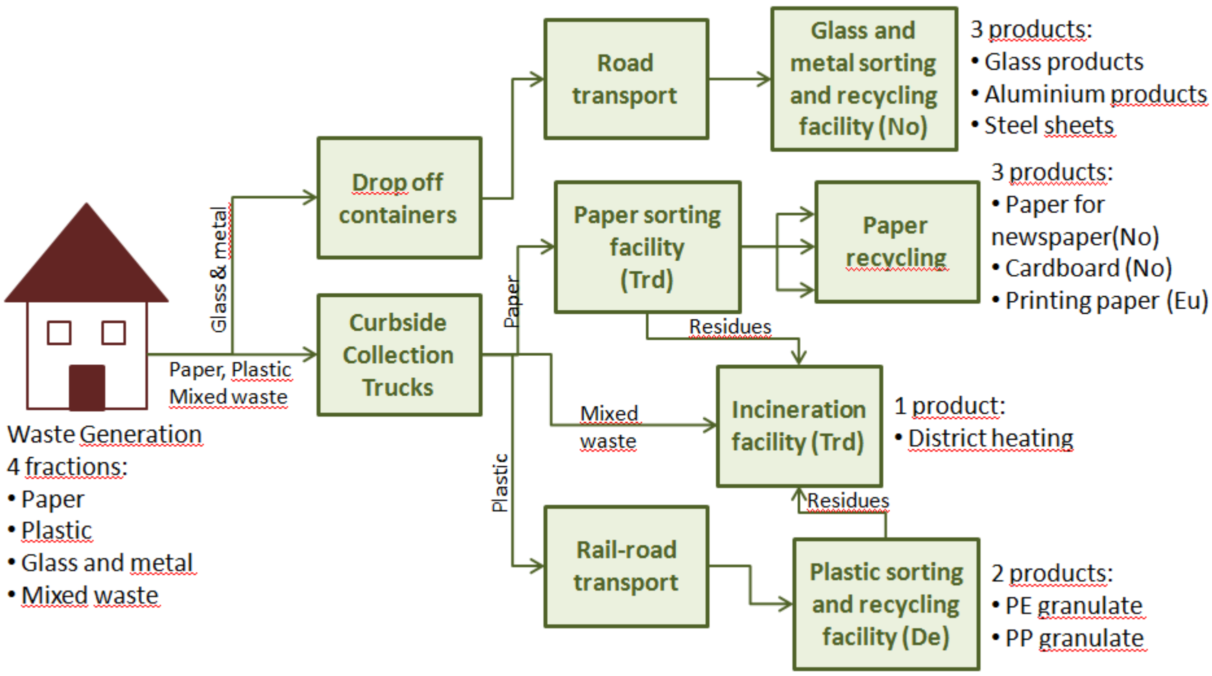


Figure 5: simplified flowchart of Trondheim's WMS

In the household waste we distinguish four fractions that are pre-sorted by the consumer and collected separately by the municipality, either via drop-off containers or by collection trucks going directly to the households. The unsorted waste, the mixed waste fraction goes then directly to Trondheim's incineration plant where it is burnt and district heating is recovered from it. Heat is supplied to Trondheim's municipality. The sorted fraction goes to Trondheim's sorting facility. From there, the plastic fraction is directly sent to Germany where it is sorted and recycled. The paper fraction is sorted in Trondheim's sorting facility and then sent to three different recycling facilities, two in Trondheim's region for newspapers and cardboard production and one abroad for printing paper production. The glass and metal fraction directly goes to some recycling facilities in south of Norway, where glass, steel and aluminium are recovered for further use. The system takes into account all kinds of transportation from one facility to another.

Residues from paper and plastic sorting are incinerated. In the system, they are sent to Trondheim's incineration plant. In practice though, residues from the German facility are obviously not going back to Trondheim to be incinerated. It was simply assumed that the incineration plant they are sent to is similar to the one in Trondheim.

As can be seen the system also takes into account substitution of virgin material production thanks to recycling and energy production via energy recovery at the incineration plant. Heat from the incinerator replaces other heating sources: wood chips heating, electricity and energy based on fuel oil and natural gas. The benefits of reduced need for virgin material and virgin energy production are thus incorporated into the waste management system. This lowers considerably the environmental impacts generated by the system. This approach is coherent with the concepts of consequential LCA, presented in the literature review and in more details in Ekvall and Weidema's article (2004).

2.3.2. Waste composition

The waste composition was determined using three different sources. First the Norwegian Statistics website was used to obtain the total amounts of main waste streams: residual waste, paper waste, plastic waste, glass and metal waste (Statistics Norway, 2012). This data is specific to Trondheim and its household waste generation. The amounts were then divided by the number of inhabitants (170936 inhabitants in 2010, Statistics Norway) and then multiplied by the average number of inhabitant per household (2.2 persons per household in Sør-Trøndelag in 2011, Statistics Norway): we obtain the waste generation per household, distributed on the four sorting categories in Trondheim.

However, waste composition must be adapted to the Easewaste categories, which are 48, and the above data is therefore not precise enough. In 2007, Norsas made samples to estimate the waste composition in Trondheim (Heie et al., 2007). The waste came from various districts and an average percentage for the entire city is also available. Norsas results are distributed into 42 categories, which is relatively close to the number of categories in Easewaste.

Numbers from Statistics Norway were first distributed on Norsas 42 categories according to the percentages determined by Norsas. Percentages for the glass and metal fractions were not included in Norsas' study so they were estimated by Helene Slagstad, thanks to data provided to her by Trondheim's municipality and from Østfoldforskning (2011). The translation from Norsas categories to Easewaste categories was then relatively easy. In cases where Norsas categories are more aggregated than Easewaste categories – as for example food waste, which is divided into animal food waste and vegetable food waste in Easewaste – the distribution was made as in Easewaste's pre-defined Danish waste composition (the average national composition in 2005). In the opposite case, Norsas numbers were simply added up to fit a more aggregated Easewaste category. The final numbers are found in *Appendix 3*.

2.3.3. Sorting efficiencies

Both the Norwegian statistics and the Norsas samples were also used to determine the sorting efficiencies in Trondheim. Norsas made samples of waste composition at the waste treatment facilities. Therefore it distinguishes the waste that goes to the recycling facilities from the one that goes to the incineration centre (Heie et al., 2007). Statistics Norway also gives the total amounts of plastic, paper, glass and metal being sorted in Trondheim. Knowing the total amounts and the percentages of each waste category in each waste fraction, it was easy to deduce the sorting efficiencies, as for instance (see *appendix 4* for the numbers):

$$\frac{Amount\ of\ magasines\ in\ the\ Paper\ waste\ fraction}{Total\ amounts\ of\ magasines\ in\ household\ waste}$$

In *Appendix 4*, one can notice that sorting efficiencies are way from being optimized in Trondheim. The plastic fraction for example contains few plastic and many other non plastic categories in significant amounts, such as glass, soil, cat litter etc.

Once sorting efficiencies, waste composition and quantity and the WMS are all defined in Easewaste the calculations can be done. Easewaste offers different views of the results, sorted by substances or processes, as graphs or tables. Results were exported on Excel.

2.3. Limitations and uncertainties of the model

There are a number of limitations that arise when using this framework. They mainly come from combining two datasets that are very different but that is not all. The limitations presented here are intrinsic to the framework; later on the uncertainties related to the choice of the scenarios will be presented too.

2.3.1. Translation from products to waste streams

The most difficult stage in building the framework is to make the link between consumption and waste generation. In figure 3 this link is shown in dashed arrows and not in full line arrows because this connection was not made for all consumption and waste categories. In many cases the correspondence was difficult to estimate, especially because the categories were not classified the same way. A consumption category such as “kitchen and domestic ustensils” is made of various materials and would thus end up in many different waste categories (“non-recyclable plastic”, “other metal”, “ceramics”, “non-recyclable glass”etc.); and it is practically impossible to know which percentages would go into each waste stream as a result of kitchen and domestic ustensils consumption.

To fix partly that problem, the best way would be to make new samplings of the waste collected in order to determin its composition in Trondheim and classify them in categories that are closer to the consumption categories.

Sometimes though, the translation from products to waste streams is not possible. Some of what ends up in the municipal solid waste is not visible in the consumer expenditure survey, simply because consumers do not pay for it, at least not directly. This is the case for advertising that is delivered in mail boxes. But the opposite situation is even more frequent: many consumption categories do not become part of the household waste. Many consumption categories are for example services, and since these categories do not generate household waste, there is therefore no connection that can be made in our framework. And even amongst material goods, some of them are treated differently when they reach the WMS. All electronic equipments, cars, but also furniture and some other large objects, are not sent to the same facilities and were not included in the waste management LCA model. They are not counted as household waste.

There are unfortunately an important number of connections that cannot be made because of that. We would first need to have a model of the entire municipal solid waste treatment system, and not only the household waste system.

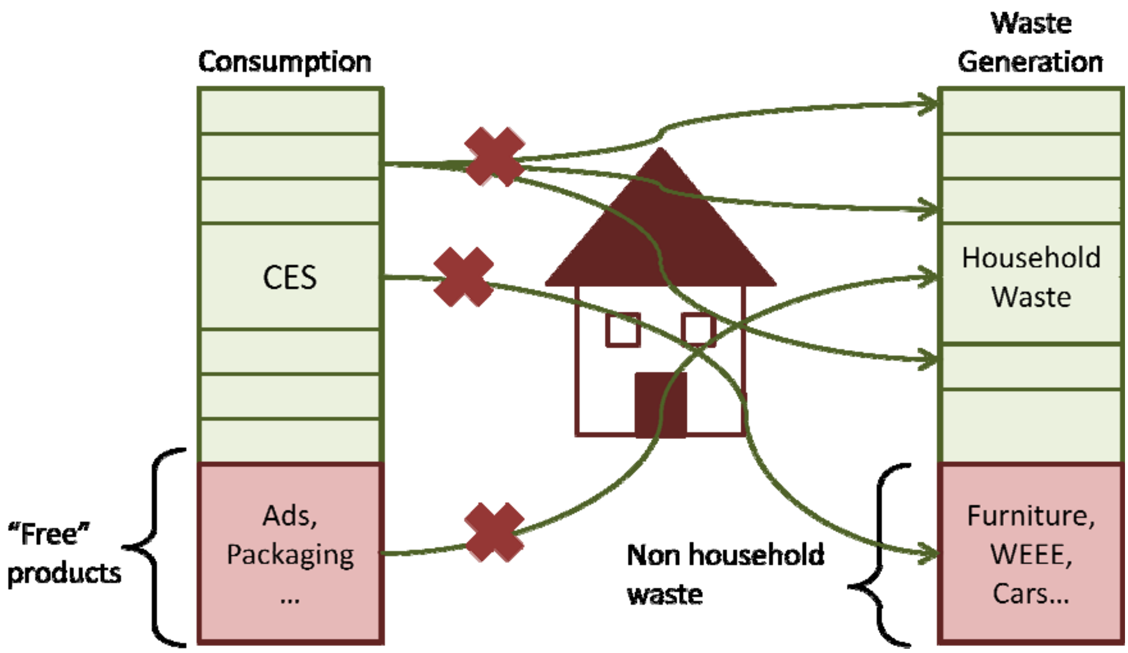


Figure 6: incomplete connection between the Consumer Expenditure Survey and the Waste Generation Vector

2.3.2. Aggregated data

Another limitation when selecting WPAs to study is the number of categories in each dataset. The consumer expenditure survey has 182 consumption categories, which is not a lot for representing the entire household consumption. The data on waste has only 48 categories. This aggregation level is a problem when trying to get more into detail about what is happening during a WPA. For example if one wants to study the benefits of reducing food waste, one should take the packaging amounts into consideration. In terms of volume, food packaging obviously represents a great part of household waste. But the waste categories in Easewaste do not separate food packaging from other packaging items. Moreover, packaging is not even visible in the CES categories. Packaging is embedded in consumption categories and is not a good in itself, which is understandable because consumers do not buy packaging alone. So there is no possible way to dissociate food packaging from food products.

More generally, this aggregation level allows making only very crude scenarios. For instance, there is no possibility of comparing benefits of organic food versus regular food, greenhouse food versus imported food, or more energy efficient washing machines versus regular ones, as it was suggested in Nonhebel and Moll (2001).

Another problem comes from the monetary units used in the IO data. In the CES the amounts purchased are implicitly linked to a physical quantity. 20NOK is the average price of 1kg of bread for example: All products have a price that is constant in the IO dataset. Therefore we cannot have a scenario where people would buy better more expensive clothes for example, and see the influence it would have on the income effect.

More important, the IO dataset does not actually provide a price vector, which is a vector which, for each product category, gives the equivalence between monetary units and physical units (20NOK/kg of bread…). For example, if we do not know the average price of one new garment and the average price of repairing one garment, how can one estimate the differences in expenses as a result of reduced garment purchase and increased repair? In one of our scenarios it was assumed that clothes purchases were reduced by 50% and that the savings made were entirely distributed on the clothes repair sector. Because of the lack of price vector it is hard to know if this represents a constraining scenario for the consumer (he or she owns substantially less clothes because the repair sector is too expensive to repair all of them) or not (the money placed on the repair sector was enough to ensure most of the clothes owned to be repaired). This will be discussed later in the next chapters. The same problem arises if one wanted to reduce meat consumption: people then need to eat more of the other food products. But how much more? And how do we estimate the amounts if we do not have the correspondence between prices and quantities?

Besides the absence of a price vector, making a vegetarian scenario faces another issue. The CES categories are quite detailed when it comes to food, but the IO categories in which they are converted are much less. Indeed there is one single category called “food products”, which aggregate all kinds of processed cereals like bread and pasta, along with jam, dried fruits, eggs, spices etc.

2.3.3. Unclear correspondence

Finally, datasets not only have different product categories, they also have different environmental stressors and environmental impacts. The Input Output data has 401 different stressors, but many of them are not the same as the ones used in the solid waste LCA. They mostly refer to different kinds of energy use, land use, domestic extraction, social and economical stressors such as employment and value added; whereas LCA stressors in Easewaste are all chemicals released in different compartments. The Input Output data has actually only 28 stressors that are actual chemicals. And they are only emissions to the air. The LCA dataset has 265 different emissions released into freshwater, seawater, groundwater, soil, air etc.

As a result some impact categories have just nothing to do with each other and are simply not comparable: there is no equivalent to the “water consumption” or “domestic extraction use” impact categories in the solid waste LCA method. Some categories look similar but are either broader or more restrictive and therefore unfortunately non-comparable. For example the IO dataset has a “Human Toxicity” impact category whereas the LCA dataset has three called “Human Toxicity via Water”, “Human Toxicity via Air” and “Human Toxicity via Soil” with different units. Some others impacts were not calculated using the same guidelines, and hence have different numbers, or even different units. For example, this is the case for the Eutrophication Potential calculated either in kg NO3-eq or in kg PO4-eq.

Finally some impact categories in the IO dataset are simply not completed yet, because the dataset is still quite recent. This was the case for radiation, abiotic depletion and ozone layer depletion.

In the end, only three impact categories were comparable from one model to another: global warming potential, acidification potential and photochemical oxidation potential. And given the limited number of common stressors, and especially the limited number of IO stressors, this would tend to underestimate the environmental impact of the upstream value chain. In practice though, this last observation is not of great importance because the uptream environmental impacts will appear to be much higher than the downstream impacts (see results). Such an influence would therefore be negligible.

All of the above limitations make it very constraining to select appropriate case studies of WPAs that impact both datasets and have clear correspondence between product category and waste streams. The following chapter starts by explaining how the case studies where chosen.

2.3.4. Rebound effect at the waste generation level

Several types of rebound effects were tested on household consumption. Rebound effects might also influence waste generation when saved money is re-spent on categories generating household waste. However, this was difficult to estimate because of the limitations already presented above: the absence of price vector, a very partial connection between CES and WGV and a WMS including only household waste and not total municipal waste. It was decided that there would be too much uncertainty in trying to do so: the rebound effect was only tested on the consumption side.

However, we have to be aware that, if the rebound effect manages to change significantly the upstream impacts, there is a chance that is modifies also significantly the waste generation. Fortunately this is of minor importance because the WMS impacts would anyway remain much smaller than the impacts from consumption (see results 4.1.1).

Besides, spending the money saved on services that do not generate household waste but that do generate municipal waste might be misleading. For example if the money saved is spent on restaurants, how can we know that there will be real waste prevention, that is to say that the amounts prevented at the household level will not be compensated by the waste generated by the restaurant? If one wanted to study the importance of a rebound effect at the waste management level, it would be necessary to have a waste management model that includes the entire municipal waste.

2.3.5. The CES-to-Y matrix

The CES-to-Y matrix can be a significant source of uncertainty in this model. It relates the CES, which is 182 categories and the IO demand vector, which is 129 categories. Sometimes the connections are quite straightforward, as for example 100% of “bananas” go to “vegetables, fruits and nuts”. Some other times though, the connection is less obvious: for example, “Other preserved or processed meat and meat preparations” is a CES category that is linked to the “products of meat cattle”, “products of meat pigs”, “products of meat poultry” and “meat products NEC” IO categories. But knowing which exactly the proportions are is highly uncertain. The CES-to-Y matrix was made manually and without a systematic approach. All connections were input one by one and case by case.

Another source of uncertainty appears in the CES vector and the CES-to-Y matrix. A consumption category was added there, to account for the fact that the CES does not include all expenditures. Indeed it was estimated that there is systematic under-reporting from the survey respondents. For instance people are less likely to buy a car during the two weeks when they are supposed to keep track of their expenses and purchases. Another example is that there will typically be under-reporting of expenditures for things like funeral services, because people do not feel like participating in a survey when one of their relatives just passed away.

In the end, this added category represents a high share of the total yearly purchase: more than 37 000 NOK, which represents no less than 10% of the expenditure. It is connected to a broad spectrum of IO categories: “meat products”, “fish products”, “radio, television and communication equipment”, “other transport equipments”, etc. Estimating the amount and especially the composition of something that was not actually bought by the household is extremely subject to uncertainties.

2.3.6. The Consumer Expenditure Survey

The comparison between the average Norwegian CES and the CES common to Trondheim, Bergen and Oslo (*appendix 1*) shows some similarities and also some important differences. The major tools and equipment category goes to more than 1900% difference. Other big differences are seen at the transportation level, with people from the three big cities spending their money much less on personal transportation than on other transport services. When looking at the categories targeted by the scenarios (food, clothing and paper), differences are smaller, but still exist. The biggest differences are around 20% in these categories.

The CES chosen is therefore not really representative of the typical Trondheim consumption pattern. This can be a significant source of biais which would not affect all scenarios equally: differences in the targeted categories are relatively similar but still not the same; and rebound effects studied affect other categories whose expenses differ substancially: the holiday rebound affects transportation and the marginal and simple rebound affect all categories. The Marginal Propensities to Spend are also calculated using average national CESs; the marginal rebound scenarios are thus concerned twice by this uncertainty.

In the end, results from the IOA should be considered much more representative of the average Norwegian household than the average Trondheim household.

2.3.7. Other sources of uncertainties

Other minor uncertainties are due to the various sources of data:

- **Different years**

Though most of the data is quite recent, there are still some differences in the years it was collected. Waste composition estimated by Norsas is from 2007, whereas household waste generation (absolute numbers) are from 2011. The CES is from 2007 to 2009 and data for the waste management model was collected from 2010 to 2012. Besides the WMS modelled on Easewaste sometimes uses pre-defined processes that are less recent. For example, the steel recycling process used was a pre-defined Danish process which is from 1992. However, it was determined that this process was still quite similar to the steel recycling facility in Norway. The Danish waste composition used to fill the gaps when adapting the waste composition to Easewaste was from 2005 (see 2.3.2). This probably results in some minor uncertainties.

- **Different geographic areas**

The data is also representative of different areas. Waste generation and composition are from Trondheim, the number of person per household is for the Sør Trøndelag region, the CES is a Norwegian average and as seen earlier some data used in Easewaste are not Norwegian but Danish. When making the scenarios many similar uncertainties occurred as well; the food waste scenario used data from the UK for example. This will be dealt with when discussing the validity of the results (see 4.2.2).

3. Case Studies

3.1. Case studies chosen

Case studies were chosen while taking into account the limitations presented above. It was important to make sure that the uncertainties were not too big to invalidate the results. Case studies therefore had to be chosen carefully.

3.1.1. Scenarios

As stated above, all case studies are both preventing household waste and reducing household consumption, one of the goals being to compare the two sides. All scenarios are also made so that they are applicable locally in Trondheim or in a particular district in Trondheim. This way we avoid extra environmental impacts generated by transport for example. Besides the data we have is on Trondheim's waste generation and composition and is hence not applicable if the waste generation occurs somewhere else.

The scenarios chosen are presented in the table below:

Table 1: list of scenarios

Waste Category	WPA type	Scenario	#
		Reference scenario: Trondheim average waste generation	0
Food	Reduce	Avoid 50% of food waste	1.0
	Shift	Shift to a more vegetarian diet (based on Alfredsson, 2004)	5.0
Textiles	Reduce	Avoid 50% of textile waste	2.0
	Reuse	Use 50% second-hand textiles (via second hand store)	2.4
	Repair	Repair 50% of old textiles (via repair store)	2.5
Paper	Reduce	Avoid 50% of paper waste	3.0
	Reuse	Share and use second-hand 50% of paper products (via libraries and shared subscription with neighbours)	3.5
All	Reduce	All 3 reduction WPAs together	4.0

In order to make the scenarios comparable with each other they must have some common points and some differences, regarding the consumption or waste categories and the WPA types for example. The common reduction percentage (50%) was determined so that it would be at the same time realistic and significant. These aspects will be discussed in the following paragraphs (see 3.1.2, 3.1.3 and 3.1.4).

The point of view chosen for elaborating the scenario was the waste generation one: a decrease in waste generation (namely waste prevention) "results" in a decrease in consumption, as opposed to a reduced consumption results in decreased waste generation. This does not change anything when a direct correspondence between the two vectors is assumed (see 2.1.4): a reduction in textile waste by 50% means a reduction in textile consumption by 50% and vice versa. However, a 50% reduction of food waste does not mean a 50% reduction in food consumption! The correspondence between consumption and waste categories and the assumptions made on that topic are detailed in paragraph 3.1.4.

3.1.2. Consumption categories

Previous studies showed that mobility, housing and food are the most important in terms of household environmental impacts (see literature review). However, since our scenarios are about preventing household waste, housing and mobility are less relevant: they mostly do not generate household waste. Waste categories were chosen so that they would be easily linkable to consumption categories (see 3.1.4) and that also represented a significant part of the household waste. The share of food waste is 25% by weight, non-packaging paper 25%, and textiles 3% (see *appendix 3* for the waste composition). The other significant waste fractions are mostly packaging, which was hard to link with consumption categories (see 3.1.4).

The choice of the waste categories investigated strongly affects the choice of the WPA types. This is related to their life span:

Table 2: Characteristics of consumption categories

Categories	Life span	Options
Food	Disposable	Reduction + Substitution
Textiles	Durable	Reduction + Reuse + Repair + Substitution
Paper	Semi-durable	Reduction + Reuse + Substitution

- **The case of food**

As a disposable good, food is not subject to reuse or repair, only to reduction or substitution. Substitution can be done by a change in diet: becoming vegetarian, vegan or simply reducing animal food consumption, but also eating more organic food and buying local and seasonal products. Given the highly aggregated data of the IO tables, one unfortunately cannot distinguish organic, local or seasonal food. We will therefore focus on reducing meat and other animal-based food waste and therefore consumption as well.

A more vegetarian diet does not *a priori* reduce the total amounts of food consumed or wasted, but it increases the eco-efficiency (amount of food per environmental impact) since animals do not have to be fed industrially to be eaten afterwards and vegetable food goes directly to the (human) consumer: there is a shift from one type of waste that is more emission intensive to another less emission intensive. Substitution is also theoretically possible in the other consumption categories (cotton instead of synthetic clothes, recycled paper instead of virgin one…) but the levels of detail in both the CES and the IOT are too low to study them.

Reduction of food related waste (food and its packaging) can have several sources: purchasing less packaging intensive products, avoiding food waste at home but also changes in diet (if people eat less they might also probably waste less food). However the aim of this study was not to get into considerations about whether the average Norwegian household should eat less or not as regard to its health. Reduction through packaging is practically impossible to assess with the given aggregated IO data: packaging of any kind is simply not visible in the IO table. The direct avoidance of food waste hence remains the best option: people eat the same amount of food but they waste less and therefore buy less. There is an effective reduction in consumption in terms of amounts purchased but not in terms of intake.

- **Paper**

Paper products (excluding thus packaging) are considered at semi-durable in the sense that they can be reused but they are not enough valuable and too fragile to be worth repairing (at least most of the time). Books, magazines and newspapers are all reusable, but magazines and newspapers to a lower extent than books since they are dependants on day-to-day news: their life span is necessarily shorter because their functionality is limited in time. Books are more likely to be reused in libraries, whereas magazines and newspapers can be reused via a more informal sharing system, at the neighbourhood level.

Office paper, advertising and other paper products are not as reusable, at least not outside the household. One can care about writing on both sides of a paper and thus “reuse” it but that would be more considered as “reduction”. Therefore the reduction and reuse scenarios will distinguished these kinds of paper products: all paper products can have their waste reduced, but only books, magazines and newspaper can be reduced.

- **Textiles**

As durable and more valuable goods, textiles have the advantage of being repairable. The repair sector exists although it costs money which sometimes makes this option not profitable for the consumer. Purchasing better quality and thus more expensive clothes would probably make their repair more desirable. However the data does not allow us to make that distinction. It can still be assumed that repairing occurs, the difficulty is then to estimate the amount of money spent on it (see 3.1.5 and 3.2).

There are also several options for exchanging second-hand clothes. They can be sold much more easily than books (books in library are generally free to use). Hence, it will be assumed that clothes reuse is made via a typical second-hand store where people donate their old clothes and that sells them at reduced prices. We do not want to consider the case of charities exporting to developing countries, as it was made in Farrant et al. 2010, because, as stated earlier, the consumers are in Trondheim. The money involved in paying for second-hand clothes is also interesting to consider since it will affect the rebound effect.

3.1.3. WPA types

After the category affected, the WPA types will also have a significant influence on the scenarios' results. Reduction, reuse, sharing or repair all succeed in decreasing total consumption and waste generation. The scenarios were defined so that they all decrease them the same way: repair, reuse or pure reduction of textiles all reduce purchases of new clothes and waste generation by 50%. However they differ in the way saved money is re-spent, and also in the level of constraint it involves for the household. Paying for second-hand goods or getting them for free and sharing purchases with neighbours (for example buying commonly a newspaper yearly subscription) result in different savings and therefore a different rebound effect. Pure reduction is likely to be more constraining for the consumers than reuse or repair. It also generates the largest savings. The following table summarises these ideas. Here we distinguish for the first time expenditure (money related) and physical consumption (physical amounts of goods excl. services):

Table 3: effects and consequences of WPA types

Options	Effects	Consequences
Reduction	Decrease in expenditure Decrease in consumption Perceived decrease in standard of living	Strong rebound effect Constraining option
Price sharing or free reuse	Decrease in expenditure No decrease in consumption Smaller perceived decrease standard of living	Strong rebound effect Less constraining option
Paid reuse	Smaller decrease in expenditure No decrease in consumption Smaller perceived decrease standard of living	Smaller rebound effect Less constraining option
Repair	Small or No decrease in expenditure Decrease in consumption (excl. services) Smaller perceived decrease standard of living	Possibly no significant rebound effect Less constraining option

Reduction was presented as the highest form of waste prevention by Cleary (2010) because it theoretically reduces the quantity of a waste stream with no transfer to other waste streams: physical consumption is reduced. Yet, it is also the option that is the most likely to be rejected by the consumer, since it involves reduced consumption without any alternative option. It is also the one that might generate the strongest rebound effect, since reduced consumption is coupled with reduced expenditure: its benefits may be offset because of this.

Price sharing or free reuse offers advantages to the consumer who can still consume in the same proportion (buy the same amounts of clothes, read the same magazines) and pays less for it. However the value of goods has changed and consumers may still perceive a decrease and their standard of leaving because they prefer new clothes to old ones and not having to share their newspapers. But it is still less constraining than pure reduction where consumers simply have less clothes and less books! However, reduced expenditure may also generate a strong rebound effect and offset environmental benefits of this option.

Paid reuse has the same effect as free reuse in the sense that the level of physical consumption stays the same. The decrease of expenditure is however smaller since second-hand goods are still purchased, though it is at lower prices. As a result a smaller rebound effect is expected.

Increased **repair** aims at reducing the total amount of clothes bought, old or new. There is hence a decrease in physical consumption. On the other hand reparable clothes might be more expensive because of better quality, as discussed earlier, and the consumer must pay for the repair service.

Having labour-intensive goods thanks to repair makes clothes less emission intensive and that was one of the options mentioned by the Perspective Project (Nonhebel and Moll, 2001) and Carlsson-Kanyama et al. (2005). Repair would result in very low (if not any) savings, especially because the price of labour is high in Norway, therefore preventing a significant rebound effect. Besides, this option adds more subjective value to goods and the Perspective Project showed that the consumers enjoyed owning good quality and expensive products.

Reuse and repair can be seen as forms of **substitution** since one replaces new clothes by old clothes or by the repair service. Substitution from a product to a related service will also be considered when taking into account the rebound effect (see 3.1.5). Dematerialization involving electronic equipments (for example reading newspapers on the internet) was not considered because electronic waste was not part of the WMS and besides estimations would have to be made about electricity use.

The effect of reduction and free reuse can appear to be exactly the same. In the first case one household decreases its consumption of new products; in the second case two or more households share common products. In both cases it is seen as a reduction in consumption in the average CES: there is no apparent difference in the scenarios' numbers. However they are not perceived the same way by consumers. Yet, since the participation rate from households was not considered, reduction and free reuse of clothes are under the same scenario 3.0. They are separated for the paper (3.0 and 3.5) only because they involve different categories (see table 4 in 3.1.4).

We should here underline the fact that the common reduction number (50%) has hence not the same "meaning" for all scenarios: since WPA options do not have the same level of constraint, 50% reduction does not result in the same effort for the consumer. Besides, there are physical limits of how much can be reused, repaired or reduced. The average household represents an average behaviour of the entire city. Hence it cannot possibly purchase 100% reused clothes: some people have to buy new clothes so that they can be reused later. There has to be enough second-hand clothes to fulfil the demand. Clothes cannot be endlessly repaired. There is an even more obvious limit to reduction: people still need to get dressed. 50% seemed to be a realistic number for all scenarios but one has to keep in mind that it is probably more constraining for the textile scenarios than for the food waste or even for the paper scenario. 50% is the maximum limit for reusing clothes without reaching shortage, assuming that all clothes are reusable once and only once.

3.1.4. Correspondence Goods – Waste streams

The following table shows the correspondence between waste categories and consumption categories for all scenarios (scenario 4.0 is simply the sum of scenarios 1.0, 2.0 and 3.0):

Table 4: correspondence between consumption and waste categories for the different scenarios

Consumption categories affected (CES)	Waste categories affected (Easewaste)	#
01 Food and non-alcoholic beverages	Animal food waste - Vegetable food waste - Paper and cardboard containers - Milk cartons - Juice cartons - Soft plastic - Clear, Green and brown glass - Aluminium foil and containers - Food cans	1.0

01 Food and non-alcoholic beverages	Animal food waste - Vegetable food waste	5.0
0312 Garments - 0313 Other clothing articles - 0321 Footwear - 052 household textiles	Textiles - Shoes – leather	2.0
0951 Books - 0952 Newspapers and periodicals - 0953 Miscellaneous printed matter - 0954 Stationery and drawing materials	Newspapers - Magazines - Books and phone books - Advertisements - Office paper - Other clean paper	3.0
0951 Books - 0952 Newspapers and periodicals	Newspapers - Magazines - Books and phone books	3.5

The connection between the two datasets is quite straightforward for the three waste streams selected. However some minor uncertainties remain: "phone books" may not be included in the CES categories, "Other clothing articles" may not include only textiles but also some plastic or metal, and finally we cannot be certain to have included all food packaging waste categories. Other more important difficulties were raised in the process of making the Goods – Waste correspondence:

- **Advertisements**

As stated earlier, advertisements are not included in the CES because consumers do not directly pay for it. However advertising in mail boxes and door front represent a large share of the household waste (6% of total weight) and it is important to include it. Hence it has been included in the waste paper scenarios, but unfortunately not in the paper consumption scenarios.

- **The packaging issue**

A similar problem arises when trying to include packaging waste in scenarios. Significant amounts of packaging are embedded especially in food consumption that is the most packaging intensive category. It was thus important to obtain estimations of the amounts (and composition) of packaging per unit of food and incorporate that in the food waste scenario. As a result preventing food waste will also result in prevention of paper, plastic, metal and glass waste. See 3.2.2 for numbers and sources.

- **Special case with food waste**

The link between food waste and food consumption is not straightforward as in the other categories. This is because most of what is bought does not end up as solid waste. Unlike any other consumption product, people really "consume" food and only the part that was not consumed ends up as waste. When we want to reduce food waste we do not want to reduce food consumption per se but we want to improve its efficiency. But making it more efficient will eventually result in a reduction in purchases. In order to connect food purchase and food waste, it was necessary to find estimations of the percentage of wasted food in the literature. Based on this review it was deduced that reducing food waste by 50% would lead to decreasing food purchase by 11%. See 3.2.1 for explanations and Quested and Johnson, 2009.

The connection between food consumption categories and food waste was problematic since there are only to food waste categories and so many on the consumption side. Fortunately, the consequences of it are not significant because all food waste goes in the end to the incinerator and burns the same way. Regarding scenario 1.0 (reducing unnecessary food waste), the waste prevented was simply assumed to be 50% animal food waste and 50% vegetable food waste. Regarding the vegetarian scenario 5.0, it was assumed that 50% of the animal food waste would become vegetable food waste without any change in total weight, which is more or less compliant with Alfredsson's new diet composition.

- **Other assumptions related to the "good – waste" equivalence**

Another issue about making a direct connection between consumption and waste generation is that sometimes we do not know for sure that the waste really goes to the household WMS. The percentage of food waste relative to food purchased (presented in the previous paragraph) stands for waste entering the household WMS and thus excludes composting, food given to animals and food waste from retail: this percentage is hence applicable to our case. However the textile scenarios are more problematic: Farrant et al. (2010) estimated that only 60% of the clothes collected for reuse are actually reused. The remaining 40% enters the municipal WMS, which does not contain only household waste.

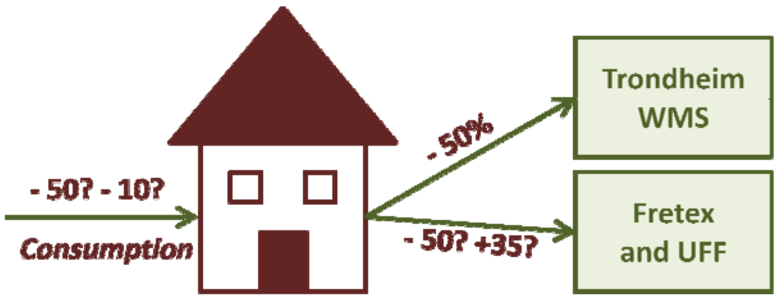


Figure 7: the future of textiles

In Trondheim, 810 tonnes of textiles were collected separately in 2011 (Statistics Norway, 2012), and, thanks to Norsas samplings, it was estimated that 22.6 kg/household/year were discarded in non-sorted household waste, that is to say 1779 tonnes in total. Therefore 31% of textiles are sorted for reuse and most of this fraction will not enter Trondheim's WMS. A significant part of it is exported (60% of it for the part collected by Fretex (2012) and 87% when collected by UFF (2012)). Since there is more than one output to textile consumption (see figure on the right) nothing tells us that a 50% reduction in textile waste going to the WMS will indeed result in a 50% reduction in new textiles purchases. It was however assumed that this is the case: that is to say that the 50% reduction of directly discarded textiles is followed by a 50% percent reduction in textiles collected for reuse and therefore a 50% reduction in consumption. A more detailed analysis would have to incorporate the second-hand market in the system under study, since it is clearly affected by our scenario.

3.1.5. Rebound effects

Various rebound effects were tested on the reduction scenarios. On the other scenarios, when savings were generated (reuse cases), the marginal rebound was applied (see 2.2.3).

The rebound effects tested are the following (the "affected" categories are the ones on which the money saved is spent):

Table 5: list of rebound effects

Rebound effect	Scenarios affected	Affected consumption categories
Simple rebound	1.1 food waste - 2.1 textile - 3.1 paper - 4.1 all	All except reduced categories
Marginal rebound	1.2 food waste - 2.2 textile - 3.2 paper - 4.2 all	All including reduced categories
Rebound on holidays	1.3 food waste - 2.3 textile - 3.3 paper - 4.3 all	073 Transport services - 096 Package holiday
Rebound on restaurant	1.4 food waste - 5.1 vegetarian diet	11111 Restaurants - 11112 Cafes, bars and the like
Rebound on outdoor culture	3.4 paper - 4.4 all	0941 Recreational and sporting services - 09421 Cinemas, theatres, concerts - 09422 Museums, zoological gardens and the like
Rebound on repair	2.5 textile	0314 Cleaning, repair and hire of clothing 0322 Repair and hire of footwear

1.0, 2.0, 3.0, 4.0 and 5.0, introduced earlier, are the corresponding scenarios without rebound. These scenarios can be seen as a rebound case where money is spent on a fictitious category that does not have any environmental burden. Donating the money to an environmental NGO may be the closest we can get to a no-burden category. It could also be a case where the household voluntarily reduces its income, by working less and earning less. Income reduction can also be involuntary, in case of economic crisis and de-growth.

The simple rebound and the marginal rebound have already been explained in the previous chapter. The rebound on repair is actually the one that has been applied to the textile repair scenario. The rest are rebound on one particular service. The service has been chosen so that it provides a similar function to those of the goods whose consumption has been reduced: restaurants and cafes for food related scenarios and cinema, museums and other outdoor activities as “replacement” for books and magazines. Regarding culture and restaurant rebounds, the aim was also to distribute the money savings on less emission intensive categories, as it was done by Carlsson-Kanyama et al. (2005).

The choice of the “holiday” rebound on the other hand was meant to show a case of rebound on a highly emission intensive category. Mobility has been highlighted in the literature (Peters and Hertwich, 2005, Carlsson-Kanyama et al, 2005, Moll et al., 2005 etc.) to be one of the most carbon-intensive activities and vacation heavily relies on mobility. Besides, “Purchase of vehicles” and “Operation of personal transport equipment” are both sets of categories that have a high Marginal Propensity to Spend in the CES (see *Appendix 2*). “Purchase of vehicles” has actually the highest MPS (13%) and “Operation of personal transport equipment” the fourth highest out of 45 (9.4%). They are hence the categories whose consumption will increase in priority in case of an increase in income. It is therefore likely that not carefully advised households participating in WPAs would spend their savings on these two sets of categories. Hopefully, better advised household would prefer to spend them on culture and restaurants. According to Alfredsson (2004), who also studied the MPSs, Swedish households would spend their extra money first on travel, then on recreation, followed by food (and then clothes, housing, services, furniture and health).

3.2. The inventory

In this chapter the data that was used to build the scenarios is presented. Some are based on literature and previous studies from Norway or other countries. Some are simply arbitrary assumptions.

3.2.1. The food waste scenario

Quested and Johnson (2009) estimated the amount of food waste that could be prevented in UK households. In the UK, food and drink waste represents 22% of the amount of food purchased per household. Avoiding food and drink waste by 50% would therefore result in consumption reduced by 11%. As for the textile scenario, there is more than one output to this consumption: most food waste goes to the WMS whereas most drink waste ends up in the sewer. Other minor “disposal” routs are home composting and feeding animals. Again it was assumed that the 50% reduction applies evenly on all of these outputs. Hence a 50% reduction of food waste in the WMS results in an 11% reduction of total food and non-alcoholic drink consumption.

Quested and Johnson (2009) estimated that the amounts of collected food and drink waste entering the WMS as household waste could be prevented by 60%. If one includes the amounts that can be “possibly” avoided (see literature review 1.4.2 for definition) then the percentage is 79%. The choice of 50% hence represents a realistic goal for the household. Household might have to adopt a more careful behaviour when purchasing and storing food but it will not involve a deep change in their consumption pattern.

At the waste generation level, the 50% reduction goal were distributed evenly on the two food categories “vegetable food waste” and “animal food waste”. At the consumption level the 11% reduction was also distributed evenly on all 62 food categories. It is thus assumed that all kinds of food are wasted in the same proportions, which is in reality not the case: Quested and Johnson (2009) estimated that fresh vegetables and salads were the most frequently wasted and next (in descending order) drinks, fresh fruits, bakery, leftover meals, meat and fish etc. It would have been possible to use Quested and Johnson data to make a more precise scenario, but given the high aggregation level of the IO dataset especially when it comes to food items, such level of detail was considered unnecessary.

- **Including packaging**

The amount of packaging waste that would be avoided as a result of food waste prevention was included in the food waste scenario. The amounts of packaging at the consumer level per tonne of food were provided by Hanne Møller who works at Østfoldforskning, a Norwegian national research company. They calculated that the total packaging system accounted for 76 kg per 1000 kg product in Norway in 2011. Among that, 45.5kg/tonne of product was the amount of packaging reaching households. The proportion of cardboard is 4.8kg, drink cardboard 7.6kg, plastic 24.3kg, glass 3.2kg, aluminum 5.5kg, and steel 0.2kg. These numbers can seem surprising because they are small and food packaging seems to be a major part of what constitutes our garbage. But though it occupies a large volume, packaging is still a very light waste.

Based on that, the amount of packaging waste was reduced proportionally to the amount of food waste prevented. For 1kg of food waste prevented, the clear, green and brown glass categories were reduced (in total) by 3.2g, the aluminium foil and containers were reduced by 5.5g etc. When there was more than one category for a material, as for the glass case, distribution was made proportionally to the initial amounts.

3.2.2. The paper waste scenarios

In the paper waste scenarios the correspondance between waste and consumption categories was more straightforward. A 50% reduction of magasines and newspaper waste results in a 50% reduction in these consumption categories. The reuse scenario was distinguished from the simple reduction scenario because it involved fewer categories: “office paper” and “other clean paper” were assumed to be non-reusable categories and therefore not included in the reuse scenario. One could argue that we can still “reuse” paper by writing on both sides of a paper sheet for example but this was considered to be a kind of reduction, because it involves a more careful personal use of paper and not a common sharing with other household.

- **the issue of advertising**

Salhofer et al. (2008) estimated that advertising paper waste represents 28 kg/cap/yr in Vienna. This includes advertising material, without any specific address, which is placed into mailboxes or hung onto doors, mail advertising which is addressed to a member of the household, advertising circulars included in newspapers and magazines as well as advertising folders added to bills. It was estimated that placing a “No thanks” sticker on the household’s mailbox would prevent 16.5 kg/cap/yr of unaddressed advertising, that is to say 59% of total advertising paper. Reducing by 50% the advertising paper waste in scenario 3.0 would therefore be a realistic assumption. Unfortunately, as stated earlier, the IO side of advertising was not taken into account because there was no corresponding category in the CES.

3.2.3. The textile scenarios

The reduce textile scenario is simple: all kinds of textiles, shoes and leather waste are reduced by 50% as well as the purchase of clothes, footwear and household textiles (bed sheets, towels etc.). The reuse and repair cases should be however developed more.

- **The reuse scenario**

Farrant et al. (2010) estimated that the proportion of collected used clothes being actually reused would be around 60%. In Norway, as seen earlier, clothes going through the Fretex and UFF systems are mainly going abroad to be reused. In the reuse scenario it was assumed that 50% of clothes purchased were second-hand clothes; hence given the small amounts available via Fretex and UFF this would not be realistic at Trondheim's level. This confirms that the choice of 50% reduction is probably more constraining in this scenario than for food or paper waste.

In the reuse scenario it was assumed that second-hand clothes and shoes were 50% cheaper than their new version. Therefore the household re-spends only 50% of the savings on them. The rest will have to be redistributed with a rebound effect (here the marginal rebound was used).

Since there is no consumption category corresponding to reused clothes and shoes in the CES, the money was artificially spent on a fictitious category that has no environmental burden. It is hence assumed that the process of reusing, which in this case relates to second-hand stores and collection from drop-off containers, has negligible impacts compared to the entire clothes and shoes production chain. Indeed the impact of making new clothes is much higher than the impact of "making" second-hand clothes (that is to say making them available to consumers). This is very likely a reasonable assumption. Local collection and a simple store do not stand against international transportation, material extraction and multiple production facilities.

The interest of having the textile reuse scenario is to evaluate the influence of a partial rebound effect: only a part of the money is re-spent on not burden-free categories. The actual price of second-hand goods does not really matter and is probably very variable, but it will affect the scope of the rebound effect: the cheapest second-hand clothes are, the largest the rebound will be.

- **The repair scenario**

The repair scenario is simply a kind of rebound effect from scenario 2.0. Money saved by reducing consumption is totally re-spent evenly on the clothes and shoes repair categories. As stated earlier, since no price vector is available it is hard to know what this scenario truly represents: if the money spent on repair is enough to repair all new clothes or not. Besides it was not possible to assume better quality and more expensive clothes since the price of goods is constant in the IO dataset.

3.2.4. The vegetarian scenario

Data from Alfredsson's PhD thesis (2002) was used to build this scenario (table 16, p.86). The diet Alfredsson investigated was first introduced by the Swedish Environmental Protection Agency. It is not a fully vegetarian diet and it follows health recommendations as well as environmental ones. Alfredsson presents both the green diet and the current Swedish diet and compares the intake per day of each food category (in grams per day).

It was first assumed that the "current" Swedish diet is similar to the average Norwegian diet and that changes towards a green diet would hence be the same. For each food category a ratio that expresses a relative change (superior to 1 in case of increase in consumption, inferior to 1 in the opposite case) can be calculated using Alredsson's comparison. Since they are unit-less, these ratios were applicable on the CES expenses. Despite a different classification of food categories, the ratios were most of the time applied on the CES categories without too much uncertainty. The correspondence between the Swedish EPA food categories and the CES was quite clear, except for the “dried leguminous plants” and the “grain and cereal” categories, which looked nothing like any category in the Norwegian CES. Unfortunately these too categories are key categories when moving to a more vegetarian diet, especially dried legumes that are the main providers of plant-based proteins. These categories are not present in the Norwegian CES probably because the quantities consumed were considered to be negligible. Luckily these two categories represent a small share of the total daily intake (in grams) and not taking them into account may not influence the results too much. But one should keep in mind that our scenario is not compliant with health requirements.

4. Results and Discussion

4.1. Results

- **The impact categories studied**

The results of the impact assessment are discussed for the three impact categories that the Easewaste and Exiopol models have in common: Global Warming Potential, Acidification Potential and Photochemical Oxidation Potential. This is so that the two sides can be compared. When looking at one side at a time, other impacts will be presented too.

Table 6: Impact potentials chosen for comparison between WMS and consumption system

Easewaste impacts	Exiopol impacts	Units	Abbreviation
Global Warming 100 Years (IPCC 2007)	Global Warming 100 Years (IPCC 2007)	kg CO2-eq	**GWP**
Acidification (EcoIndicators 95)	Acidification (fate not incl.) (Hauschild & Wenzel 1998)	kg SO2-eq	**AP**
Photochemical Ozone Formation, Low NOx (EDIP97)	photochemical oxidation (low NOx) (Andersson-Sköld et al. 1992)	kg C2H4-eq	**POCP**

- **The reference scenario**

The reference scenario, representing the current situation in Trondheim, generated the following impacts per household and per year:

Table 7: impacts generated by the reference scenario

	GWP (kg CO2-eq)	AP (kg SO2-eq)	POCP (kg C2H4-eq)
Consumption side	22746	322	4.26
Waste management side	-109	-0.919	-0.0357

From this table we can notice that, first, waste management related impacts are extremely small compared to impacts related to consumption and production and second, that these impacts are actually negative. As was stated earlier, the waste management system includes not only waste treatment and sorting but also production of material and energy from waste. The WMS works as an alternative producer of energy, via the incineration, and of material, via recycling. It substitutes virgin material and energy production and the assessment includes the benefits of this avoided production. The overall effect of the waste management system is then positive, at least for these three impacts. It is not the case for other impacts, as will be presented in 4.1.2.

In figure 8 below, the reference scenario is aggregated into the main consumption categories and the IO results are expressed in terms of share of GWP, AP, POCP and expenditure. The category "other goods" gathers all remaining goods such as alcohol and tobacco, jewellery, goods for personal care, recreational goods and communication goods. The category "other services" gathers for example education, health and cultural services. This graph will be important to understand the results in 4.1.1.

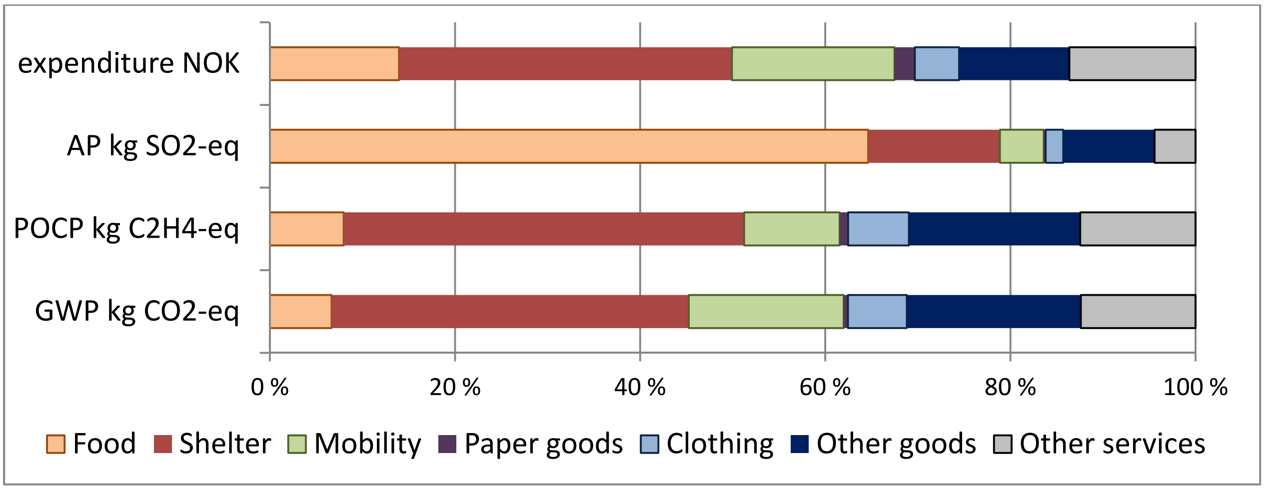
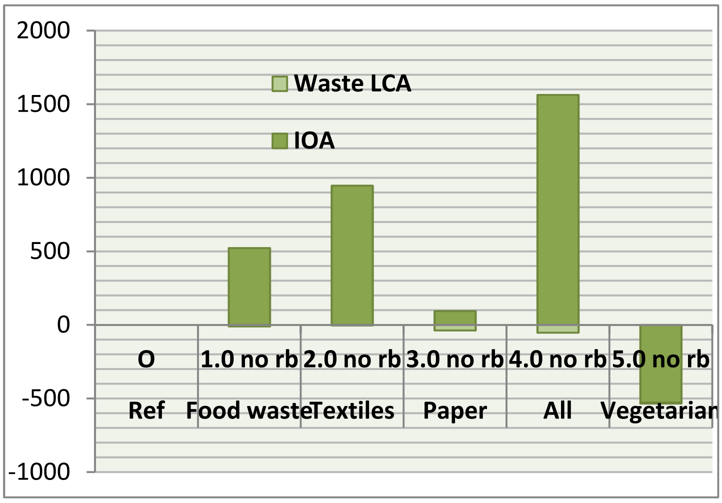


Figure 8: Share of Expenditure, GWP, AP and POCP by consumption categories in the reference scenario

4.1.1. No rebound scenarios, upstream and downstream results

In the following graphs, the results are presented relative to the reference scenario, for both the waste and the consumption sides. A positive value means that the scenario generates savings in the concerned impact category compared to the reference scenario. A negative value means that the scenario generated more environmental damage than the reference scenario.

Figure 9 shows the total change in Global Warming Potential generated by each scenario. Most scenarios achieve a net reduction in GWP, which was the intended purpose. Scenario 4.0 generates as much savings as the three first scenarios combined, which is not surprising since it was designed that way.



A first surprise is the increased GWP generated by the **vegetarian scenario**. Vegetarian diets have been highlighted in the literature as having a significant potential for reducing environmental impact and our scenario shows us the opposite. Moreover, the data used was extracted from Alfredsson's work and her no-rebound scenario did generate some environmental benefits.

Figure 9: GWP savings for both upstream and downstream, numbers relative to reference scenario, all scenarios without rebound

The problem lies in the way the IO dataset is structured. Indeed, there seem to be a high detail level in the CES when it comes to waste: the CES has 62 food categories, which represents no less than 33% of the CES categories. However, the IO tables have much fewer food categories: only 11 out of 129 categories. When making the connection between the CES and the IO tables, one had to aggregate many categories together and these grouped categories were then assigned the same emission coefficients. As a result, many different types of food products end up in the “Food products NEC” category in the IO table: bread, pasta, pastries, but also berries, eggs, dried fruits, chocolate, jam, sauces and other kinds of processed food. And all of these food products grouped together thus have the same environmental impact per Euro of product.

In the vegetarian scenario, consumption of bread and pasta was highly increased. And when comparing the “food products” category to other food categories, such as different kinds of meat, processed rice and vegetables, it has the highest carbon intensity (here expressed in GWP/million Euro). That could explain the significant increase in GWP compared to the reference scenario. To conclude, the current state of the IO dataset is not detailed enough to build a proper vegetarian scenario.

Secondly, figure 8 can be used to understand the **difference between the food waste, textile and paper scenarios**. In the reference scenario, the GWP associated to food consumption is slightly bigger than the one associated to clothes and footwear, but this difference is small: in the end, food consumption is reduced by only 11% while textiles are reduced by 50% and this is what makes the clothes contribution more important. The difference between AP associated to food and clothes is on the other hand much higher: AP related to food consumption represents more than 60% of the total AP. That is why, in figure 10, the food waste scenario remains the most preferable, despite the 11% reduction.

Paper products are sub-categories of “recreation and culture” a represent a very small part of the total expenditure (2%, while food represents 10% and clothes 4%). Their share of the three environmental impacts related to consumption is also very small (0.4% of GWP, 0.2% of AP and 0.9% of POCP). It is therefore understandable that reduced paper consumption does not achieve as much benefits as food and clothes reduction. The exception is found in figure 11 for POCP, where paper achieves higher benefits than food on the IO side. The explanation for the results on figure 11 is presented in the next page.

A third important remark on figure 9 is that **the waste LCA side shows very little changes** compared to the IOA side. This confirms again what was found in the literature. Several authors (Ekvall 2008, Olofsson 2004) estimated that the benefits of waste prevention would mostly occur because of the avoided production resulting from reduced consumption. Besides, in figure 9 one observes not only small environmental benefits from the WMS, but actually reduced benefits: since less recyclable or valuable waste via energy recovery is brought to the WMS, less recycled material and energy is available to substitute virgin production. The effect of waste prevention on the WMS is seen as negative.

Figure 9, 10 and 11 show that the environmental impact from the WMS is negligible compared to the benefits of reduced consumption, except for the paper scenario. As far as paper is concerned, the benefits on the IO side are small compared to the textile and food scenarios, at least for GWP and AP; and they are partly compensated by the reduced benefits of paper recycling. Reduced paper purchases generate 93kg CO_2-eq while reduced paper waste takes back -36kg CO_2-eq. Benefits of reduced consumption of paper are also partially offset in the Acidification Potential in **figure 10**.

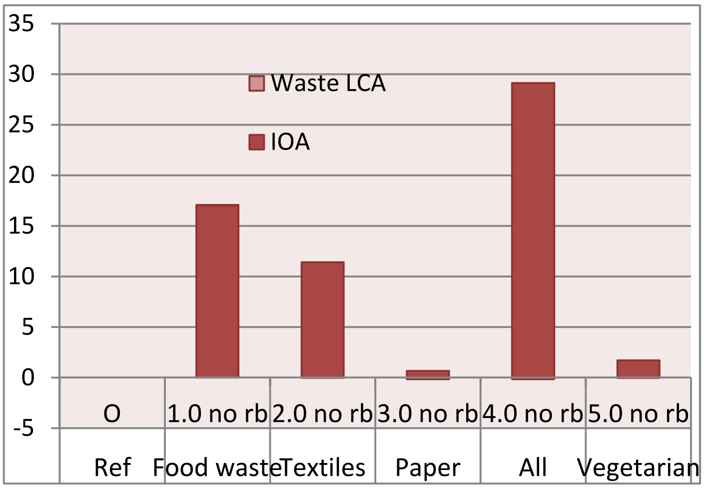


Figure 10: AP for both upstream and downstream, numbers relative to reference scenario, all scenarios without rebound

Figure 10 shows similar trends to figure 8: benefits of the vegetarian diet are not negative but still very low; the paper scenario shows still very small effects, both upstream and downstream. And food and textile scenarios succeed in generating significant upstream benefits, unless this time food waste wins the first place (for the reason explained above).

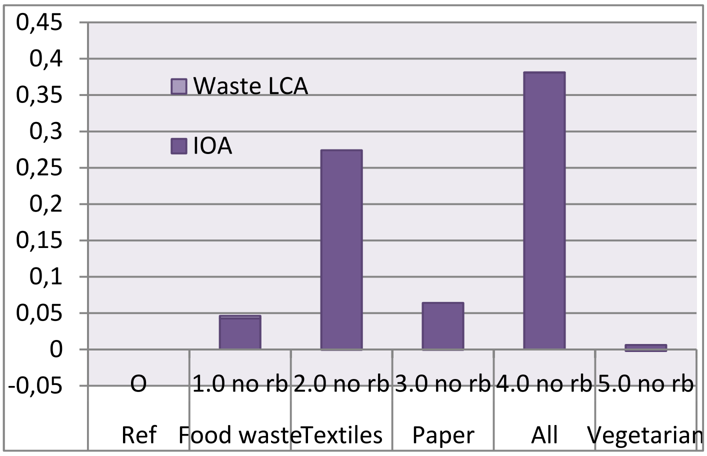


Figure 11: POCP for both upstream and downstream, numbers relative to reference scenario, all scenarios without rebound

In **Figure 11**, the ranks of the food, textile and paper scenarios are different from the two other impact categories. Figure 8 does not really help understanding the reasons for this. To explain these results, one needs to dig in a bit deeper in the IO dataset.

POCP has two related emissions according to the characterisation matrix: methane and carbon monoxide. When looking quickly at the stressor matrix, we notice that the values for these emissions follow the same pattern as figure 11: “printed matter” generates around 25kg C2H4-eq/Euro, “manufacturing of textiles” 100 kg C2H4-eq/Euro and “food products” only 9kg C2H4-eq/Euro. When multiplying by the reduction in expenditure in each scenario, the results are still complying with figure 11. These approximate numbers are valid when assuming that most paper and food production occur in Europe, while textile production occurs in the developing world. According to the Norwegian household demand vector, this is a reasonable simplification.

4.1.2. Waste management system

The impacts generated by the WMS were not easy to visualise in the previous graphs since they were so small compared to the IO assessment. The following graphs show only the WMS results and present them as percentages of savings compared to the reference scenario. To explain the graphs it is first necessary to keep in mind the most contributing processes to the three impacts. The following table summarises the values of five main processes and their contribution to the reference scenario. The last line represents the impacts for the entire WMS, including the five processes and 46 others.

Table 8: main contributing processes to the reference scenarios. in kg CO2-eq, kg SO2-eq and kg C2H4-eq respectively

	GWP	AP	POCP
Newspaper and magazines to Newspaper, Stora Enso, Sweden, 2008	-57	-0,33	-0.0074
Incineration, grate furnace, Trondheim, N, March 2011	-6.6	-0,24	-0,021
Glass cullets (colored) to new products (100% recycled), S, 2008	-10.5	-0,08	-0.055
Aluminium scrap to new products (remelting), Europe, 1993	-16.5	-0.14	0.036
Steel scrap to steel sheets, DK, 1992	-24	-0.08	-0.011
Total	**-109**	**-0.92**	**-0.036**

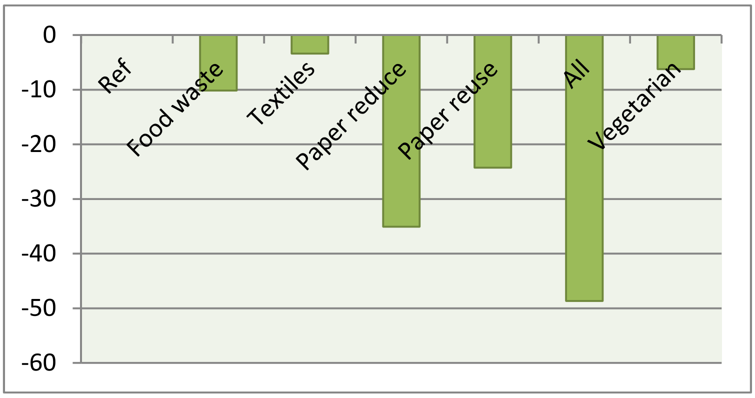


Figure 12: GWP savings (percentages), waste LCA, all aggregated scenarios

In **figure 12** we see that all scenarios generated less environmental benefits than the reference scenario. The **paper scenarios** are the ones that are the least beneficial. This can be explained by the fact that the paper recycling processes contribute a lot to a reduced GWP (see table 8 above). Paper reduction involves both the recycling of newspapers and magazines and the recycling of other kinds of paper. The latter are less contributing processes (-6.4 and -3.5 kg CO2-eq) than the former (-57 kg CO2-eq) but that is what makes the difference between the two paper scenarios: paper reuse only involves newspaper and magazine waste.

Food waste and textile waste are all going to the incinerator which is a less contributing process than the recycling of paper. The reason why food waste prevention is less beneficial than textile waste prevention is that the food waste scenario involves larger amounts: food waste represents 25% by weight of the entire waste generation; hence reducing it by 50% is much more effective than reducing textile waste by the same percentage. In the food waste scenario, the benefits of recycling paper and metal packaging appear to be not significant for GWP, because they represent very small amounts.

The only difference between the reference scenario and the **vegetarian scenario** was that half of the animal food waste was turned into vegetable food waste. The reason why the vegetarian scenario generates fewer benefits than the reference scenario is that the heating value of animal food waste is higher than the one of vegetable food waste: 24.55 GJ/ton against 18.3 GJ/ton. Animal food waste combustion hence generates more energy.

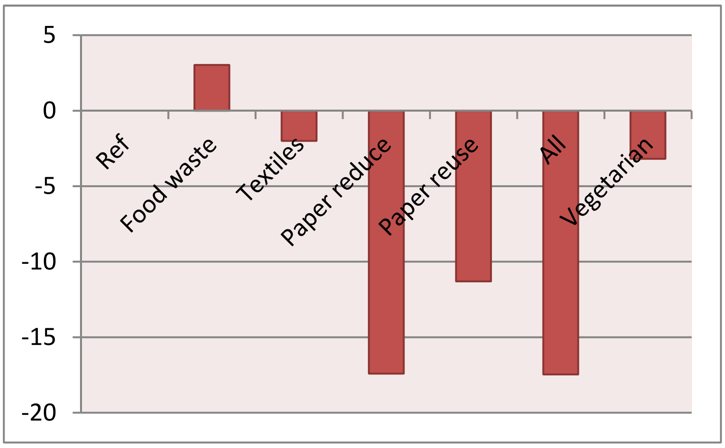


Figure 13: AP savings (percentages), waste LCA, all aggregated scenarios

In **figure 13** the trend for the paper scenarios is reinforced since the newspaper recycling process is even more important for AP than for GWP: the newspaper recycling represents more than one third of the total Acidification gains from the reference scenario (see table 8 above).

The food waste scenario manages to generate benefits compared to the reference scenario. The reason lies in the direct emissions occurring at the incinerator level: the "process specific emissions" in the incinerator are decreased because much less waste is burnt. The textile scenario involves much smaller amounts, and in the vegetarian scenario the total amount of food is kept unchanged and therefore this reason is not applicable. The vegetarian scenario is worse than the reference scenario again because of a lower heating value.

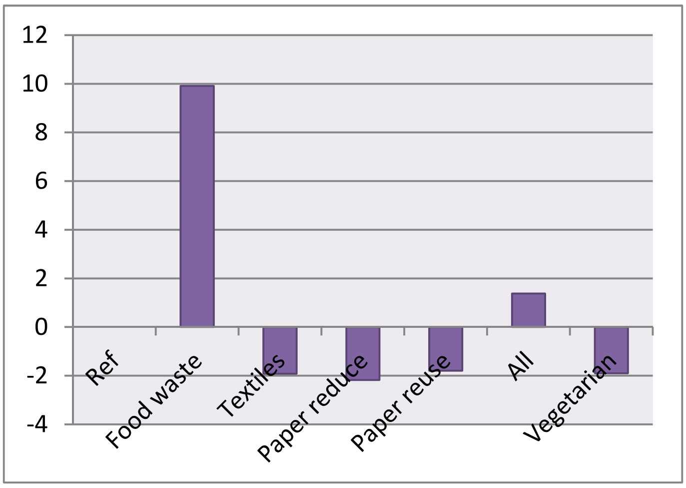


Figure 14: POCP savings (percentages), waste LCA, all aggregated scenarios

In **figure 14**, the food waste scenario generates clear benefits. Surprisingly, this is due to the reduced need for aluminium recycling, which is a process that contributes negatively a lot to POCP (see table 8 above). The food waste scenario reduces the aluminium packaging waste by a very small amount but which is enough to change the overall POCP significantly.

In the vegetarian scenario, no change in packaging was considered so this scenario does not generate benefits. Overall, the influence of the WMS on the POCP is small, compared to the two other impact categories, where the overall scenario 4.0 reaches almost -18% in AP and -50% in GWP. As a result, very small processes become important in the final result, such as aluminium recycling in the case of the food scenario.

- **Other environmental impacts**

GWP, AP and POCP generated by the WMS are very small, but it might not be the case for other environmental impacts. The following table presents other impact categories and compares the case of Trondheim's WMS to an open dumping that would gather the same amount and composition of waste.

Table 9: Comparison between Trondheim's WMS and an open dumping, other impact categories

	Human Toxicity via Water (EDIP97): [m3 water]	Spoiled Groundwater Resources: [m3 spoiled water]	Eco-toxicity in Soil (EDIP97): [m3 soil]	Human Toxicity via Air (EDIP97): [m3 air]	Stored Ecotoxicity in Water (EDIP): [m3 water]	Human Toxicity via Soil (EDIP97): [m3 soil]	Stored Ecotoxicity in Soil (EDIP): [m3 soil]	Ecotoxicity in Water, Chronic (EDIP97): [m3 water]
reference	1629	1.87	-65.2	9.10^{7}	3832420	-20.6	7.59	-111734
open dumping	42.4	4405	33.5	10^{8}	4227220	4.69	15.6	4723

Not surprisingly, most of the time Trondheim's WMS is preferable to an open dumping. The exception is for "human toxicity via water", where the reference scenario is much worst. Sometimes though, even if Trondheim's WMS performs better than the open dumping, impacts are no longer negative and seem to be significant, as it is the case for "human toxicity via air" and "stored ecotoxicity in water". Since we have no impact categories in the IO dataset to compare them with, it is currently not possible to evaluate whether these three impacts are significant compared to the overall life cycle impacts. Further attempts to compare upstream and downstream household impacts should consider them.

4.1.3. Rebound effects on the upstream value chains

In 4.1.1 we already saw the results for scenarios without rebound effect. Now it is time to compare the influence of the different rebound effects tested. The following graphs express environmental benefits in percentages, comparing them to the reference scenario.

As expected, all rebound effects tend to limit the initial benefits made from reduced consumption. In any case money was re-spent on categories that have an environmental burden. However, one can observe significant differences from on rebound to another. Looking at **figure 15**, representing GWP, some of the expectations we had were confirmed: the holiday rebound is the worst one as it manages to offset the most part of the benefits from scenario 4.0; the rebound on culture is the most preferable one as scenario 4.4 still manages to reach more than 5% GWP savings; the simple and marginal rebounds are most of the time quite similar.

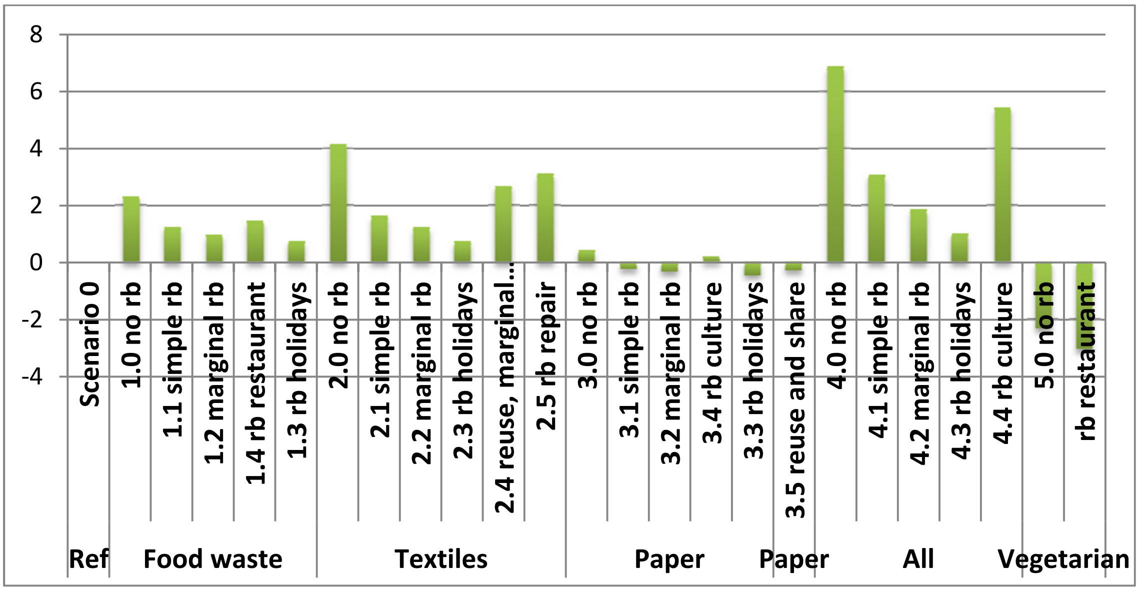


Figure 15: GWP savings (percentages), IOA, all scenarios

The simple rebound is more optimistic than the marginal rebound. This was also expected since the simple rebound is more of an objective rebound whereas the marginal rebound takes into account consumers preferences. And as said earlier, these preferences are mainly set on "Purchase of vehicles" and "Operation of personal transport equipment", which are carbon intensive categories. The marginal rebound scenarios lie in the middle between the simple rebound scenarios and the holiday rebound scenarios.

The restaurant rebound is preferable to the simple rebound and seems to be relatively close to the culture rebound. They both have low carbon intensity. The rebound on repair also has a similar effect. The reuse textile scenario offers a middle way between no rebound a total marginal rebound.

Fortunately, most of the time, the rebound effects do not completely offset environmental benefits. However they do for the paper scenarios. Only the culture rebound manages to still generate some benefits (even very little). The other rebound effects make it not desirable to undertake paper waste prevention. Not surprisingly, the vegetarian scenario got worst when a rebound was added.

Figure 16 shows that the rebound effects do not have the same rank for AP as they had for GWP. The culture rebound is still the most preferable, but now the holiday rebound comes second and the rebound on restaurant is now no better than the simple rebound. This probably comes from the fact that food is such a big contributor to Acidification Potential, and restaurants, unlike travelling, involve high amounts of food. Textile repair remains a desirable rebound effect and textile reuse is still a middle way between no rebound and total rebound.

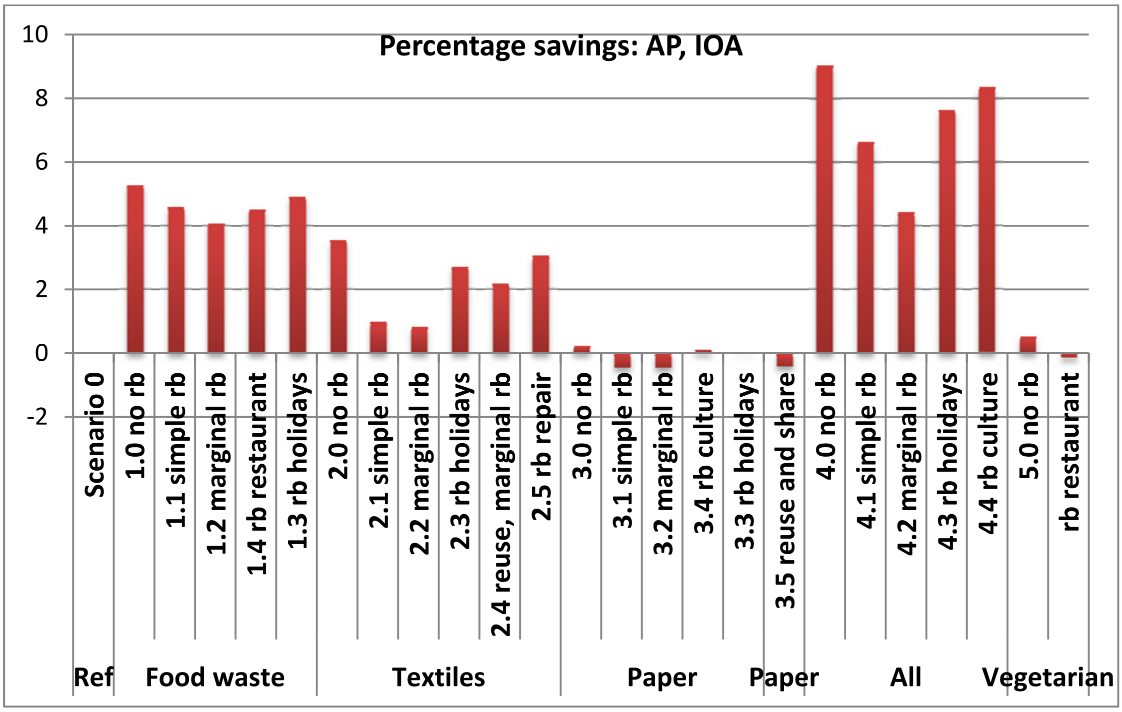


Figure 16: AP savings (percentages), IOA, all scenarios

The marginal rebound is still less beneficial than the simple rebound, probably again because food has a high MPS (it is the third highest): in the marginal rebound there is hence a significant amount of money that is spent on food.

Figure 17 shows the results for Photochemical Oxidation. Rebounds on culture, holidays, restaurant and repair are all showing higher benefits than the simple and the marginal rebounds and they are all quite similar. The marginal rebound is again worse than the simple rebound.

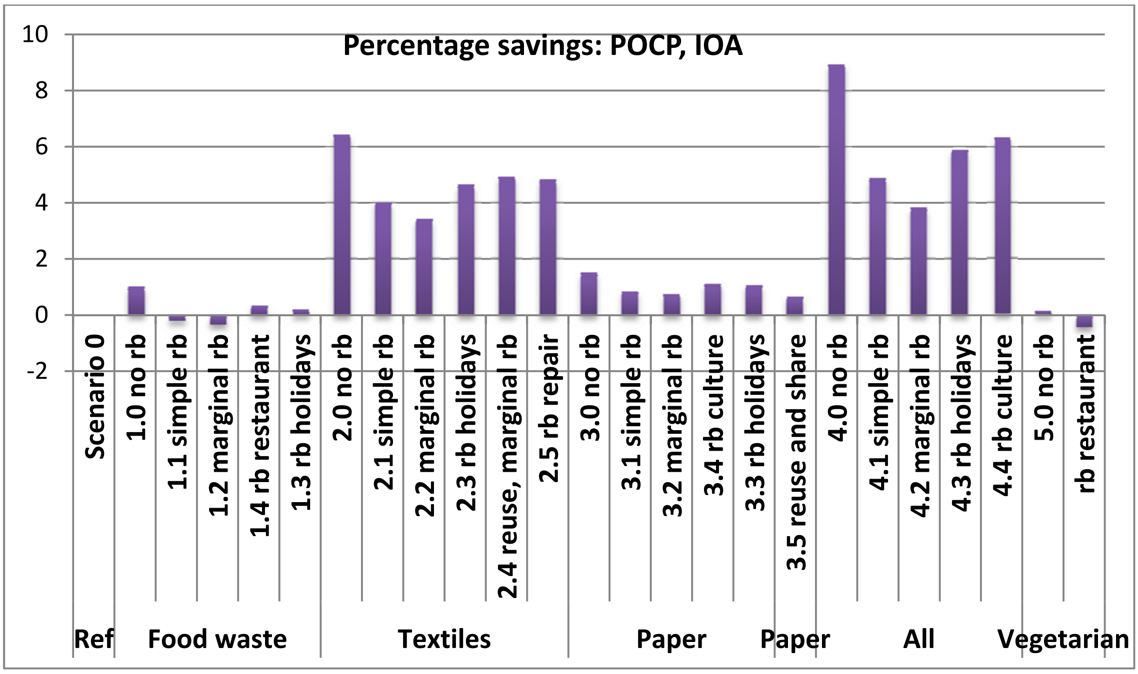


Figure 17: POCP savings (percentages), IOA, all scenarios

4.2. Discussion

4.2.1. Comment on the results

- **The general result**

Overall the results show relatively low environmental benefits. The best case scenario 4.0 barely reaches a 7% reduction of GWP and approximately 9% of AP and POCP. Scenario 4.0 is besides not the most realistic scenario since it assumes no rebound. If a rebound is included, then the benefits stand mostly between 3 to 6%. Carlsson-Kanyama et al. (2005) estimated that total energy use related to household consumption could be reduced by 10 to 20%.

However, the present scenarios have been including only a small part of the total consumer expenditures (about 20%) and this part represents a small share of the total GHG emissions (approximately 14%). On the GWP side, higher benefits could probably be reached by including more carbon intensive categories such as transport and housing. Carlsson-Kanyama et al. considered all aspects of consumption. By looking only at the consumption categories generating household waste, the total environmental benefits that could be made were restricted. Electronic and mechanical equipments contribute probably a lot to indirect emissions but were not included because they were not counted as household waste. Housing electricity and heat consumption, as well as the direct emissions of mobility are simply not material goods and cannot be counted as “waste”.

- **The scenarios**

Looking at the different scenarios, reduction of textile purchases seems to have the most positive impact. But one has to keep in mind that the food scenario involved only 11% reduction of food purchases. If this percentage was higher, then the food scenario would maybe outweigh the textile scenario. We saw that the textile scenario was probably quite constraining for the consumer; if the reduction percentage was decreased to 30% for example, then the textile scenario may not be the best one.

It is unfortunate that the vegetarian scenario did not give conclusive results; otherwise it could have been added to scenario 4.0 and succeeded in increasing benefits significantly. The values for the paper scenario are so low that one could deduce that paper waste prevention should not be undertaken. However, one should not draw such conclusion: this only means that paper waste prevention should not be undertaken alone (otherwise any rebound effect would offset it) and that it should not be the main priority.

- **The WMS**

The three WMS related impacts presented above are negative and very small compared to the consumption side impacts. This would tend to support the idea that there is no need for increased detail on the waste management system. Looking at the general IO tables may be enough. Again it should be underlined that other important impacts such as human toxicity and ecotoxicity were not compared between the two sides, and the conclusion of such comparison could differ. Besides the waste composition does not include the disposal of some problematic products such as electronic equipments being discarded as WEEE (Waste of Electronic Electric Equipment) which might generate much more impact than the typical household waste.

Another question is: would the WMS still be negligible if it was based on landfilling a not energy and material recovery? The table below shows a comparison between Trondheim's WMS and an open dumping, which is probably the worst case of WMS. The open dumping scenario is receiving the same kind of waste as the WMS in the reference scenario and was made on Easewaste.

Table 10: Comparison between Trondheim's WMS and an open dumping, for GWP, AP and POCP

	GWP (kg CO2-eq)	AP (kg SO2-eq)	POCP (kg C2H4-eq)
Open dumping	464	0,109	0,242
Trondheim WMS	-109	-0.919	-0.0357

The open dumping generates impacts, unlike Trondheim's WMS that shows negative values. But the impacts generated still have the same order of magnitude as Trondheim's values. When we compare them to the upstream impact from consumption, they still represent quite a small share.

- **The rebound effects**

The rebound scenarios showed that the rebound effect can have a great influence on the final results. The holiday scenario manages to reduce almost entirely the GWP benefits from scenario 4.0. The marginal rebound is what would most likely happen if the rebound was not controlled. It shows that it is important not to focus only on a set of consumption categories but on the entire consumption pattern. The way money saved is respent should be controlled as well as the purchase on the targeted categories. Sustainable consumption will not succeed in reaching significant environmental benefits it the emissions are simply shifted from one category to another.

Rebounds on recreational services are the most preferable ones. The conclusions taken out of the literature review and especially regarding the Perspective Project (Nonhebel and Moll, 2001) and Carlsson-Kanyama et al (2005) showed that is was possible to control the fate of the total expenditure by providing people with new enjoyable lifestyles: services and high quality products are both well appreciated by consumers and are the best way to spend extra money.

4.2.2. Uncertainties and possible improvements

The various incertainties were enumerated in the previous chapter 3 and the end of the second chapter 2.3. Two kinds of uncertainties are differenciated: the ones linked to the framework itself, inherent to a particular dataset or because of the connection with different datasets, and the ones linked to the case studies chosen. The question is then, which uncertainties are the most important and are they big enough to compromise the conclusion of the results?

The two IO and LCA frameworks combined in this master thesis were already separetely in use by other researchers before this project was started, but they are both quite recent and subject to regular updates and upgrades. We have pointed out the gaps present in the characterisation matrix and the uncertainty behind the CES-to-Y matrix as important examples. For this study the stressor matrix needed to be rearranged to fit the characterization matrix, because the categories in the two matrices were not in the same order. This showed that it was probably the first time that these two matrices were used together. The WMS model is also subject to uncertainties especially regarding the treatment and recycling of the metal and glass fraction. The model has been refined very recently to represent better the reality of waste management in Trondheim, changing the recycling rate of the incinerator's bottom ash, the efficiency of the steel recycling plant etc.

The data characterising the household: its income, its consumption pattern and its waste generation is also subject to uncertainties. Statistical data, if gathered from a sufficient number of household and amount of waste, can be quite well representative of the reality. But we have seen that the national CES used did not represent well the average Trondheim household and that the IOA results could potentially be different if the proper CES had been used. The main source of error for this data, as explained earlier in 2.3.5, comes from the CES respondents themselves, and this was partly corrected by the addition of an extra consumption category (also subject to errors).

But the main uncertainties probably come from the link that was made between the three vectors, the CES, the Y demand vector and the WGV. The CES-to-Y matrix was used with no modification but we should keep in mind the high uncertainty related to it. The connection between the consumer expenditures, and the waste generation is even blurrier. The scenarios were chosen to minimize the uncertainty behind this connection. But if new scenarios were to be made, one should first collect new data to improve this part of the framework.

First, new samples of the waste composition in Trondheim could be made. It could be classified in a way that would fit the CES better: distinguishing food packaging from other kinds of packaging; instead of having a soft plastic category, distinguishing the origin of plastic waste etc. The WMS should include the recycling and reuse of beverage cans and plastic bottles, as well as the recycling of WEEE (Waste of Electric and Electronic Equipments), furniture, cars and other significant household waste streams that have been omitted in this study. The disposal of WEEE is a concerning issue for health and environmental reasons. Resource depletion is a key indicator for the various rare materials present in electronic equipments; and WEEE recycling in developing countries, where WEEE from western countries are sometimes sent, might not be done in the safest way for the employees and the local environment (Yang et al., 2008).

One should also improve the detail level of the IO dataset. Some scenarios are simply not possible to build because of too few product categories. The resolution of the food categories especially is much too low. But this represents a lot more efforts since the IO dataset gathers data from 44 countries and international trade between them. Meanwhile one could try to use the Dutch database which has a much higher resolution but which needs to be adapted to Norwegian conditions and to include the emissions related to imports and foreign technologies. Besides, the Dutch database only provides data on GWP and energy intensity. No database is perfect for the use we make of them and one has to choose what aspect is the most important for the study: the precision of the scenarios or the quality of the background data.

The aim of all of these possible improvements would be to have a higher variety of case studies and more representative of the total waste generated at the consumer level.

Finally, in order to have an idea of the reduction percentages that can be potentially accepted by the consumer, one should connect this study with a behavioural study, by making surveys for example. The main problem with the scenarios that were presented in this report is that their feasibility and their reasonability were not estimated. A questionnaire was made for the people who are at the moment considering of moving in Brøset, the future green neighbourhood in Trondheim. Unfortunately we did not optain the feedbacks from this questionnaire on time to incorporate them in the case studies. Knowing the opinion of those who are probably more environmentally aware on average than other people as regard with what they are ready to change in their personal habbits would be interesting. Not necessarily in order to know the potential participation rate of a scenario, because it would probably be much higher in Brøset than in the rest of Trondheim; but at least to have an idea of how constraining some scenarios can be, and estimating reduction percentages that would represent approximately the same efforts for the consumer.

Using surveys to make new scenarios would help estimating the real potential of sustainable consumption, not just using arbitrary reduction percentages. Textiles reduction may come first in the GWP results, but it may be because it is the most constraining scenario, and one should not deduce that textile waste prevention should be considered as a priority. One should keep in mind that the comparison between the present scenarios is not meant to attribute different levels of importance but only to inform about the potential environmental benefits of each of them. Common reduction percentages in the scenarios were not meant to be representative of what could be done in reality, and the same goes for other assumptions, such as the price of reused clothes, the price of repair, or again the rebound effects: one does not expect that money saved would be spent on only one consumption category.

More generally, given the high uncertainty of the current framework and case studies, the results are only meant to inform about the relative importance of the two sides of consumption: the production chain and the waste management. They also show the importance of the rebound effect in the final benefits and the importance of not only a category’s total emissions, but also its emission intensity, thus relating the final environmental footprint of a consumer to the money spent and the prices of products.

Conclusion

The main results from the case studies analysed are the following:

First, the environmental benefits of waste prevention occur mostly at the production chain level, which confirms what was found in the literature (Ekvall 2008, Olofsson 2004): Global Warming changes occurring at the waste management level are 0.3 to 3% of the ones occurring upstream; for POCP they are from 0.2 to 11% and 0.1 to 1% for AP. The exception is for the paper scenario, where changes in GWP at the WMS level compared to the reference scenario are 36% those of the consumption side, which is not negligible; for AP the percentage is 24% and POCP only 1%. However, all these percentages actually represent negative changes; this means, in the case of GWP, that 36% of the environmental benefits generated at the production chain level are offset by reduced performance of the WMS.

This is because the Waste Management System generates environmental benefits on its own, thanks to energy recovery and material recycling that substitute primary production. Decreasing the amounts of waste collected hence reduces these benefits. In the end, for the three impact categories investigated, almost all scenarios tend to decrease the WMS's benefits. But this decrease is most of the time negligible compared to upstream generated benefits.

Second, the results showed that the influence of the rebound effects is significant. In the case of global warming, the holiday rebound is the one that mitigates the most the initial benefits (they are reduced from 7% to 1% in the results that combine all scenarios together). Rebounds on restaurant, repair and culture are the most beneficial in the way that they reduce the benefits only from 7% to 5.5%. The marginal and the simple rebound stay in between, the marginal being worse than the simple rebound because of high MPSs on food and mobility. Regarding AP and POCP, conclusions are similar to the GWP results except for the holiday rebound which is then preferable to the marginal and simple rebound. However this rebound should not be preferred given the very bad effect it has on GWP.

Third, the comparison between the different targeted categories, food, textiles and paper, is subject to high uncertainties. Hence no final conclusion should be drawn from it. However, one can still notice the importance of food for GWP and AP, as it generated significant benefits even though food consumption was reduced by only 11%: the total GWP from household consumption was reduced by 2.3% in this scenario without rebound. The 50% textiles reduction generated twice more benefits when it comes to GWP, but one has to keep in mind that this scenario is likely to be more constraining for the consumer. The results also showed that preventing paper waste is the least beneficial scenario. Sometimes its benefits can be completely offset by the rebound effect and the negative effect at the WMS level. This however does not mean that paper waste prevention should be abandoned; it only means that it should not be undertaken alone.

Further analysis aiming at assessing the potential of both waste prevention and sustainable consumption should first improve the framework so that other important consumption categories can be included: cars, electronic equipments, furniture etc., all these categories would be interesting to investigate on both the waste management side and the production side. Efforts should be made to reduce the sources of errors present in the framework and in the choice of case studies, so that clear final conclusions can be drawn out of it.

References

Eva C. Alfredsson, 2004: *"Green" consumption – no solution to climate change*. Energy 29 (2004) 513–524

Eva C. Alfredsson, 2002: *Green consumption energy use and carbon dioxide emission.* Doctorat thesis, department of social and economic geography, spatial modelling centre, Umeå University

Arne Støttrup Andersen, 2001: *The Norwegian System of Social Surveys.*
Available online at:
http://www.mzes.uni-mannheim.de/projekte/mikrodaten/wp_pdf/wp_26_norway.pdf

Annika Carlsson-Kanyama, Rebecka Engström, Rixt Kok, 2005: *Indirect and Direct Energy Requirements of City Households in Sweden Options for Reduction, Lessons from Modeling*. Journal of Industrial Ecology, Volume 9, Number 1–2

Julian Cleary, 2010: *The incorporation of waste prevention activities into life cycle assessments of municipal solid waste management systems: methodological issues*. International Journal of Life Cycle Assessment (2010) 15:579–589

Jayne Cox, Sara Giorgi, Veronica Sharp, Kit Strange, David C. Wilson, Nick Blakey, 2010: *Household waste prevention – a review of evidence*. Waste Management & Research 2010: 28: 193–219

Jayne Cox, Brook Lyndhurst, 2009: *The evidence base for household waste prevention: selected findings from a review for Defra*. Available on:
http://www.resourcesnotwaste.org/upload/documents/webpage/RRF%20Conference%20Materials/WP(2009)/JayneCox-email.pdf

Faye Duchin, 2005: *Sustainable Consumption of Food: A Framework for Analyzing Scenarios about Changes in Diets.* Journal of Industrial Ecology Volume 9, Number 1–2

Tomas Ekvall, 2008: *Waste prevention: Environmental effects and policy instruments*. NORDIC WORKSHOP – WASTE RESOURCE MANAGEMENT AND CLIMATE ACTIONS, 10 June 2008, Fredrikstad, NORWAY

Tomas Ekvall and Bo P. Weidema, 2004: *System Boundaries and Input Data in Consequential Life Cycle Inventory Analysis*. International Journal of Life Cycle Assessment 9 (3) 161 - 171 (2004)

Exiopol, 2010: explanation of the top-down approach available on:
http://www.feem-project.net/exiopol/scheda.php?ids=45

Laura Farrant, Stig Irving Olsen, and Arne Wangel, 2010: *Environmental benefits from reusing clothes.* International Journal of Life Cycle Assessment (2010) 15:726–736

Fretex, 2012:
information available online at: http://www.fretex.no/readimage.aspx?asset=3714

Emmanuel C. Gentil, Daniele Gallo, Thomas H. Christensen, 2011: *Environmental evaluation of municipal waste prevention.* Waste Management xxx (2011) xxx–xxx

Aage Heie, Camilla Louise Bjerkli, Marius Johansen, 2007: *Renholdsverket i Trondheim Analyse av husholdningsavfall i Trondheim kommune.* Norsas, December 2007

Edgar G. Hertwich, Anders H. Strømman, 2011: Presentation by Anders Strømman at NTNU for the Industrial Ecology MSc program. *"The Rebound Effect, Marginal Expenditure and Sustainable Consumption".* Fall semester 2011.

Mikko Jalas, 2002: *A time use perspective on the materials intensity of consumption*. Ecological Economics 41 (2002) 109–123

Eléonore Lèbre, 2011: *Life cycle assessment of waste prevention activities: a review of methodologies*. Project report, non published

Brook Lyndhurst, 2011: *Consumer insight: data labels and storage guidance, final report*.
Available on: http://www.wrap.org.uk/sites/files/wrap/Technical_report_dates.pdf

Henri C. Moll, Klaas Jan Noorman, Rixt Kok, Rebecka Engström, Harald Throne-Holst, and Charlotte Clark, 2005: *Pursuing More Sustainable Consumption by Analyzing Household Metabolism in European Countries and Cities*. Journal of Industrial Ecology, Volume 9, Number 1–2

S. Nonhebel, H. C. Moll, 2001: *Evaluation of Options for Reduction of Greenhouse gas Emissions by Changes in Household Consumption Patterns*. Final report to the NRP of the GreenHouse Project. IVEM-ONDERZOEKSRAPPORT NO. 106

Mattias Olofsson, 2004: *Improving Model-Based Systems Analysis of Waste Management*. Thesis for the degree of doctor of philosophy. Department of Energy Technology, Chalmers University of Technology, Göteborg, Sweden

Michael Ornetzeder, Edgar G. Hertwich, Klaus Hubacek, Katarina Korytarova, Willi Haas, 2008: *The environmental effect of car-free housing: A case in Vienna.* Ecological Economics 65 (2008) 516 – 530.

Glen P. Peters and Edgar G. Hertwich, 2006: *The Importance of Imports for Household Environmental Impacts*. Journal of Industrial Ecology, Volume 10, Number 3

Tom Quested and Hannah Johnson, 2009: *Household Food and Drink Waste in the UK*, final report. Report prepared by WRAP. Banbury

A. H. M. E. Reinders, K. Vringer, and K. Blok. 2003: *The direct and indirect energy requirement of households in the European Union.* Energy Policy 31(2):139–153.

Stefan Salhofer, Gudrun Obersteiner, Felicitas Schneider, Sandra Lebersorger, 2008: *Potentials for the prevention of municipal solid waste*. Waste Management 28 (2008) 245–259

Helene Slagstad, Helge Brattebø, 2012: *LCA for household waste management when planning a new urban settlement*. Waste Management xxx (2012) xxx – xxx.

Statistics Norway, 2012: main page for data on consumption: http://www.ssb.no/forbruk_en/
List of tables related to consumption available on:
http://statbank.ssb.no/statistikkbanken/Default_FR.asp?Productid=05.02&PXSid=0&nvl=true&PLanguage=1&tilside=selecttable/MenuSelP.asp&SubjectCode=05
Main page on waste generation:
http://statbank.ssb.no/statistikkbanken/Default_FR.asp?Productid=01.05&PXSid=0&nvl=true&PLanguage=1&tilside=selecttable/MenuSelP.asp&SubjectCode=01
Table on Trondheim population:
http://www.ssb.no/english/subjects/02/02/folkendrhist_en/tables/tab/1601.html
Table on number of person per household:
http://www.ssb.no/english/subjects/02/01/20/familie_en/tab-2011-04-07-02-en.html

S. Suh, 2004: *Functions, commodities and environmental impacts in an ecological–economic model*, Ecological Economics Journal, 48 (2004) 451– 467

Arnold Tukker, Maurie J. Cohen, Klaus Hubacek, and Oksana Mont, 2010: *The Impacts of Household Consumption and Options for Change*. Journal of Industrial Ecology, Volume 14, Number 1

UFF, 2012: information available online at:
http://www.hias.no/Artikkel.aspx?AId=1439&back=1&MId1=41&MId2=&MId3=&

Jianxin Yang, Bin Lu, Cheng Xu, 2008: *WEEE flow and mitigating measures in China*. Waste Management Journal, Volume 28, Issue 9, 2008, Pages 1589–1597

Østfoldforskning, 2011: Publication available online at:
http://ostfoldforskning.no/publikasjon/klimaregnskap-for-avfallshandtering-fase-i-og-ii-576.aspx

Appendix

Appendix 1: Comparison between the average Norwegian Consumer Expenditure Survey and the average CES for Oslo, Bergen and Trondheim (CESs from Statistics Norway, 2012)

Consumer Expenditure Survey	Average Oslo, Bergen and Trondheim	Average Norway	% change
0111 Bread and cereals	6412	6845	0.0675
0112 Meat	7859	9202	0.170
0113 Fish	2762	2950	0.0680
0114 Milk, cheese and eggs	6366	7146	0.122
0115 Oils and fats	772	965	0.25
0116 Fruit	3365	3697	0.0986
0117 Vegetables	4236	4504	0.0632
0118 Sugar, jam, honey, chocolate and confectionery	3894	4261	0.0942
0119 Food products	2393	2485	0.0384
0121 Coffee, tea and cocoa	921	1046	0.135
0122 Mineral waters, soft drinks, fruit and vegetable juices	3362	3431	0.0205
0211 Spirits	1163	1196	0.0283
0212 Wine	3666	2921	-0.255
0213 Beer	2963	2555	-0.159
022 Tobacco	2968	3608	0.215
0311 Clothing materials	40	54	0.35
0312 Garments	17514	16369	-0.0699
0313 Other articles of clothing and clothing accessories	834	1008	0.208
0314 Cleaning, repair and hire of clothing	145	176	0.213
0321 Shoes and other footwear	3623	3242	-0.117
0322 Repair and hire of footwear	19	16	-0.187
0411 Actual rentals paid by tenants	16430	9135	-0.798
0412 Other actual rentals	878	905	0.0307
0421 Imputed rentals of owner-occupiers	78205	62835	-0.244
0422 Other imputed rentals	157	401	1.55
0431 Materials for the maintenance and repair of the dwelling	13548	18163	0.340
0432 Services for the maintenance and repair of the dwelling	8502	8788	0.0336
0441 Water supply	817	1429	0.749
0442 Refuse collection	892	1656	0.856
0443 Sewerage collection	730	1516	1.07
0444 Other services relating to the dwelling	454	550	0.211

0451 Electricity	11584	15043	0.298
0452 Gas	64	61	-0.0491
0453 Liquid fuels	923	916	-0.00764
0454 Solid fuels	671	2028	2.02
0511 Furniture and furnishings	7652	8010	0.0467
0512 Carpets and other floor coverings	441	461	0.0453
0513 Repair of furniture, furnishings and floor coverings	:	:	:
052 Household textiles	1786	2062	0.154
0531 Major household appliances whether electric or not	2356	3136	0.331
0532 Small electric household appliances	821	504	-0.628
0533 Repair of household appliances	82	60	-0.366
054 Glassware, tableware and household utensils	1884	2039	0.0822
0551 Major tools and equipment	12	248	19.6
0552 Small tools and miscellaneous accessories	1615	2283	0.413
0561 Non-durable household goods	3446	3530	0.0243
0562 Domestic services and household services	1478	968	-0.526
0611 Pharmaceutical products	2919	3244	0.111
0612 Other medical products	344	315	-0.0920
0613 Therapeutic appliances and equipment	1749	1365	-0.281
0621 Medical services	1414	1410	-0.00284
0622 Dental services	2916	2932	0.00548
0623 Paramedical services	690	529	-0.304
063 Hospital services	341	264	-0.291
0711 Motor cars	21981	31055	0.412
0712 Motor cycles	187	971	4.19
0713 Bicycles	679	767	0.129
0721 Spare parts and accessories	2058	2105	0.0228
0722 Fuels and lubricants	7731	11991	0.551
0723 Maintenance and repair of personal transport equipment	3635	5319	0.463
0724 Other services in respect of personal transport equipment	4099	2727	-0.503
0731 Passenger transport by railway	2127	1175	-0.810
0732 Passenger transport by road	4201	2493	-0.685
0733 Passenger transport by air	4847	2743	-0.767
0734 Passenger transport by sea and inland waterway	816	893	0.0943
0736 Other purchased transport services	1893	2294	0.211
081 Postal services	506	365	-0.386
082 Telephone and telefax equipment	992	1013	0.0211
083 Telephone and telefax services	6503	6191	-0.0504
0911 Equipment for the reception, recording and reproduction of sound and pictures	5013	4347	-0.153
0912 Photographic and cinematographic equipment and optical instruments	731	1075	0.470
0913 Information processing equipment	2482	2324	-0.0679

0914 Recording media	1240	1125	-0.102
0915 Repair of audio-visual, photographic and information processing equipment	11	23	1.091
0921 Major durables for outdoor recreation	2435	4684	0.923
0922 Musical instrument and majors durables for indoor recreation	365	294	-0.241
0923 Maintenance and repair of other major durables for recreation and culture	355	473	0.332
0931 Games, toys and hobbies	1672	1835	0.0974
0932 Equipment for sport, camping and open-air recreation	2277	1724	-0.320
0933 Gardens, plants and flowers	2534	2999	0.183
0934 Pets and related products	1459	2752	0.886
0941 Recreational and sporting services	4366	3159	-0.382
0942 Cultural services	5878	5378	-0.0929
0943 Games of chance	2339	2667	0.140
0951 Books	2156	1909	-0.129
0952 Newspapers and periodicals	2709	3156	0.165
0953 Miscellaneous printed matter	375	319	-0.175
0954 Stationery and drawing materials	471	369	-0.276
096 Package holiday	9284	8730	-0.0634
101 Pre-primary and primary education	107	81	-0.320
102 Secondary education	284	212	-0.339
103 Post-secondary non-tertiary education	:	:	:
104 Tertiary education	757	423	-0.789
105 Education not definable by level	222	144	-0.541
111 Catering services	16284	12294	-0.324
1111 Restaurants, cafes and the like	15287	11501	-0.329
1112 Canteens	997	794	-0.255
112 Accomodation services	942	1320	0.401
1211 Hairdressing salons and personal grooming establishments	3767	3529	-0.0674
1212 Electrical appliances for personal care	126	131	0.0396
1213 Other appliances, articles and products for personal care	5659	5203	-0.0876
1231 Jewellery, clocks and watches	1083	1349	0.245
1232 Other personal effects	1151	1212	0.0529
124 Social protection	3455	3989	0.154
1252 Insurance connected with the dwelling	2722	3725	0.368
1253 Insurance connected with health	121	278	1.29
1254 Insurance connected with transport	5548	4296	-0.291
126 Financial services	447	456	0.0201
127 Other services	820	544	-0.507

Appendix 2: Marginal Propensities to Spend

MPSs were calculated comparing CESs from households having a yearly income between 410 000 and 569 999NOK and households with an income between 570 000 and 779 999NOK.

$$MPS_i = \frac{Change\ in\ expenditures\ on\ component\ i}{Total\ change\ in\ expenditures}$$

Where the total change in expenditure is equal to 537 453 – 414 199 = 123 254 NOK. 537 453 is the average yearly expenditure for households having an income between 570 000 and 779 999 NOK. And 414 199 is the same for the other group of households.
The average Norwegian household, the one whose CES was used in the study, has an income between 410 000 and 569 999 NOK. The MPSs express the additional expenses such household would make if it was given extra money.

011 Food	**0,10811**
012 Non-alcoholic beverages	0,012251
021 Alcoholic beverages	0,012559
022 Tobacco	-0,00804
031 Clothing	0,070854
032 Footwear	0,013639
041 Actual rentals for housing	-0,02837
042 Imputed rentals for housing	0,113538
043 Maintenance and repair of the dwelling	0,03681
044 Water supply and miscellaneous services related to the dwelling	0,007635
045 Electricity, gas and other fuels	0,021955
051 Furniture and furnishings, carpets and other floor coverings	0,021557
052 Household textiles	0,010353
053 Household appliances	-0,00112
054 Glassware, tableware and household utensils	0,011732
055 Tools and equipment for house and garden	0,011334
056 Goods and services for routine household maintenance	0,015204
061 Medical products, appliances and equipment	0,007935
062 Out-patient services	0,007059
063 Hospital services	-0,00012
071 Purchase of vehicles	0,132077
072 Operation of personal transport equipment	0,094082
073 Transport services	0,041832
081 Postal services	0,001444
082 Telephone and telefax equipment	0,002523
083 Telephone and telefax services	0,004422
091 Audio-visual, photographic and information processing equipment	0,010231
092 Other major durables for recreation and culture	0,028794
093 Other recreational items and equipment, gardens and pets	0,026758

094 Recreational and cultural services	0,026904
095 Newspapers, books and stationery	0,017079
096 Package holiday	0,026409
101 Pre-primary and primary education	0,000187
102 Secondary education	0,00043
103 Post-secondary non-tertiary education	0
104 Tertiary education	-0,00032
105 Education not definable by level	-0,00029
111 Catering services	0,027618
112 Accomodation services	0,010085
121 Personal care	0,041078
123 Personal effects	0,017192
124 Social protection	0,037256
125 Insurance	0,006815
126 Financial services	0,003302
127 Other services	-0,00081
Total	**0,999968**

Source:
http://statbank.ssb.no/statistikkbanken/Default_FR.asp?Productid=05.02&PXSid=0&nvl=true&PLanguage=1&tilside=selecttable/MenuSelP.asp&SubjectCode=05

Appendix 3: Waste composition and waste amounts in Trondheim per household and per year. Presented for each waste fraction.

Total amounts (kg/hh/yr) Trondheim, average	Residual waste	Paper waste	Plastic waste	Glass waste	Metal waste	Total	Compo-sition %
Vegetable food waste	126	0	0,0733	0	0	126	18,8
Animal food waste	39,9	0	0,0230	0	0	39,9	5,93
Newsprints	17,1	64,8	0,1620	0	0	82,1	12,2
Magazines	4,70	17,7	0,0443	0	0	22,4	3,34
Advertisements	7,92	34,3	0,0688	0	0	42,3	6,29
Books, phone books	0,670	0,965	0,0109	0	0	1,64	0,245
Office paper	2,28	3,28	0,0370	0	0	5,60	0,833
Other clean paper	6,37	9,17	0,103	0	0	15,6	2,32
Paper and cardboard containers	7,92	11,7	0,0550	0	0	19,7	2,93
Other clean cardboard	13,5	6,10	0,0825	0	0	19,7	2,92
Milk cartons (carton, plastic)	4,66	2,44	0,0275	0	0	7,12	1,06
Juice cartons (carton, plastic, aluminium)	0,932	0,916	0,0275	0	0	1,87	0,279
Kitchen towels	14,257	0,071	0,0383	0	0	14,3	2,13
Dirty paper	10,8	0,054	0,0292	0	0	10,9	1,62
Dirty cardboard	5,63	0,028	0,0151	0	0	5,67	0,844
Soft plastic	39,1	0,153	4,81	0	0	44,1	6,55
Plastic bottles	0,932	0	0,0825	0	0	1,01	0,151
Hard plastic	21,9	0	4,40	0	0	26,3	3,91
Non-recyclable plastic	8,38	0	1,60	0	0	9,99	1,48
Yard waste, flowers	14,9	0	0,0138	0	0	14,9	2,21
Animal excrements and bedding (straw)	0	0	0	0	0	0	0
Diapers, sanitary towels, tampons	19,5	0	0	0	0	19,5	2,91
Cottons, bandages	0	0	0	0	0	0	0
Disposable sanitary products (cloths, gloves)	0	0	0	0	0	0	0
Wood	3,72	0	0,0275	0	0	3,75	0,558
Textiles	22,3	0	0,233	0	0	22,6	3,36
Shoes, leather	3,26	0	0,110	0	0	3,37	0,501
Rubber	0,932	0	0,0275	0	0	0,960	0,143
Plastic products (toys, hangers, pens)	0	0	0	0	0	0	0
Cigarette butts	0	0	0	0	0	0	0
Other combustribles	2,79	0	0,0275	0	0	2,82	0,419
Vacuum cleaner bags	13,8	0	0,0204	0	0	13,8	2,05
Clear glass	4,03	0	0,0413	7,89	0	11,9	1,78
Green glass	4,03	0	0,0413	7,89	0	11,9	1,78
Brown glass	4,03	0	0,0413	7,90	0	11,9	1,780

Non-recyclable glass	2,33	0	0,055	0	0	2,38	0,354
Beverage cans (aluminium)	0,46	0	0	0	0	0,466	0,069
Aluminium foil and containers	3,26	0	0,0138	0	1,73	5,01	0,745
Food cans (tinplate/steel)	5,12	0	0,0275	0	15,6	20,8	3,092
Plastic-coated aluminium foil	2,79	0	0,0275	0	0	2,82	0,419
Other metal	6,99	0,305	0,0413	0	0	7,33	1,091
Soil	1,11	0,0150	0,126	0	0	1,24	0,185
Stones, concrete	2,58	0,0352	0,295	0	0	2,91	0,432
Ash	4,81	0	0,0071	0	0	4,81	0,716
Ceramics	1,80	0,0245	0,206	0	0	2,03	0,302
Cat litter	4,46	0,0609	0,511	0	0	5,03	0,748
Batteries	0,932	0	0	0	0	0,93	0,138
Other non-combustibles	1,22	0,0167	0,140	0	0	1,38	0,206
Total	465	152	13,74	23,7	17,4	672	100

Data calculated from Norsas (Heie et al. 2007) and Statistics Norway (2012).

Appendix 4: Sorting efficiencies

Percentages of a waste category found in a given waste fraction:

In recyclable waste fraction, many different waste categories can be found. Some of them should not be there and are not even recyclable (especially in the plastic waste fraction).

Waste fraction	Plastic %	Glass & metal %	Paper %
Vegetable food waste	0,00058		
Animal food waste	0,00058		
Newsprints	0,00197		0,788
Magazines	0,00197		0,788
Advertisements	0,00163		0,811
Books, phone books	0,00661		0,586
Office paper	0,00661		0,586
Other clean paper	0,00661		0,586
Paper and cardboard containers	0,00279		0,595
Other clean cardboard	0,00419		0,309
Milk cartons (carton, plastic)	0,00386		0,342
Juice cartons (carton, plastic, aluminium)	0,0146		0,488
Kitchen towels	0,00266		0,0049
Dirty paper	0,00266		0,0049
Dirty cardboard	0,00266		0,0049
Soft plastic	0,109		0,0035
Plastic bottles	0,0813		
Hard plastic	0,167		
Non-recyclable plastic	0,161		
Yard waste, flowers	0,00092		
Wood	0,00733		
Textiles	0,01035		
Shoes, leather	0,03264		
Rubber	0,02867		
Other combustribles	0,00974		
Vacuum cleaner bags	0,00147		
Clear glass	0,00344	0,659	
Green glass	0,00344	0,659	
Brown glass	0,00344	0,659	
Non-recyclable glass	0,02307		
Aluminium foil and containers	0,00274	0,346	
Food cans (tinplate/steel)	0,00132	0,752	
Plastic-coated aluminium foil	0,00974		
Other metal	0,00563		0,0416
Soil	0,1014		0,0121
Stones, concrete	0,1014		0,0121
Ash	0,0014		

Ceramics	0,1014	0,0121
Cat litter	0,1014	0,0121
Other non-combustibles	0,1014	0,0121

Data calculated from Norsas (Heie et al. 2007) and Statistics Norway (2012).