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The Rebound Effects of Switching to Vegetarianism

A Microeconomic Analysis of Swedish Consumption Behavior

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Declaration

I hereby affirm that the present thesis with the title "The Rebound Effects of Switching to Vegetarianism" was prepared by myself alone and did not involve the use of any impermissible help or of any other tools than the ones indicated. All parts of the text – including tables, maps, figures, etc. – which were taken over verbatim or analogously from other published or unpublished works have been identified accordingly. The thesis has not yet been submitted in the same or a similar form within the context of another examination, and has not been published either in part or in its entirety.

Uppsala, July 8, 2014

J. Grabs

Janina Grabs

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My dear, dear AFEPA friends – you have been my study buddies, fika dates and travel companions, and more importantly, you have been my family for this time. I cannot express how much your friendship means to me. You all deserve the best, and I hope that your dreams will come true and that we will never lose touch.

Finally, I would like to thank my family, especially my parents, for putting up with my erratic nomad lifestyle and always encouraging me to work, travel, and explore more. Thank you for supporting me as often as my plans and my next destinations change, for bolstering my belief that I have made the right decisions, and for always providing a comforting home base between my adventures.

As the Dalai Lama said: "Give the ones you love wings to fly, roots to come back and reasons to stay." You, dear friends and family, have given me all three of those, and I am exceedingly grateful for it.

Abstract

In order to reduce our environmental footprint, policy-makers have increasingly focused on influencing individual-level consumption choices. Recent years have seen a special focus on sustainable eating patterns, in particular the environmental benefits of a vegetarian diet. However, reliable conclusions on this issue need to take full-scale behavior changes into consideration. This can be achieved using the concept of the indirect rebound effect, which describes the amount of potential environmental improvements *not* realized due to the re-spending of expenditure saved during the initial behavior shift. This study aims to quantify the potential environmental savings stemming from the shift of an average Swedish consumer to vegetarianism, as well as the most likely rebound effects, in terms of both energy use and greenhouse gas emissions. To this end, it estimates Engel curves of 117 consumption goods, derives marginal expenditure shares from them, and links these values to environmental intensity indicators. Results indicate that switching to a vegetarian diet could save an average Swedish consumer 16% of the energy use and 20% of the greenhouse gas emissions related to their food and drink consumption. However, if they re-spend the saved income according to their current preferences, they would forego 96% of the potential energy savings and 49% of the greenhouse gas emission savings. These rebound effects are even higher for lower-income consumers, since they tend to re-spend on more environmentally intensive goods. Yet, the adverse effect could be tempered by simultaneously purchasing organic goods or by re-spending the money exclusively on services. Thus, consumption advice should shift to promoting holistic sustainable lifestyle changes.

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Abbreviations

AES	Actual energy savings
EAP	Energy analysis programme
EEIO	Environmentally extended input-output (table)
GHG	Greenhouse gases
kg CO_2-eq	kilograms of CO_2-equivalent emissions
LCA	Life cycle analysis
MES	Marginal expenditure share
PES	Potential energy savings
MJ	Megajoule
R	Rebound effect
SEK	Swedish crown

Note from the author:

The use of “we” and “our” and any other similar expressions is intended for writing style purposes only. I am the sole person responsible for and author of the present thesis.

1. Introduction and Problem Statement

This chapter first introduces the study's motivation and research questions. It then provides a summary of the research hypotheses to be tested. Finally, it introduces the methods and results that will subsequently be presented in more detail in the following chapters.

1.1. Motivation and context of the study

The unsustainability of Western lifestyles has increasingly become a topic of interest to researchers and policy-makers alike. Environmental footprint analyses reveal that we would require a bio-capacity of 4.5 Earths if every human attempted to live like the average American (Pollard 2010). In response to this phenomenon, governments and civil society organizations have often called for individual-level consumption changes. In particular, nutritionally comparable dietary choices have shown to be vastly different in a range of environmental impact factors (Hertwich & Katzmayr 2003). This difference is especially apparent when comparing animal-based products to plant-based ones, and a reduction of individual consumers' meat consumption has therefore been suggested by a number of authors as a significant and relatively easy contribution to more sustainable lifestyles (Taylor 2000; Steinfeld et al. 2006; McMichael et al. 2007; Carlsson-Kanyama & González 2009; González et al. 2011). However, these conclusions are reached analyzing sector-specific consumption choices and ignoring total household expenditure behavior (Murray 2013). In particular, the indirect environmental rebound effect – the environmental effect of re-spending money saved by not purchasing meat or fish – of vegetarianism has never been investigated. This study intends to address this research gap in order to inform consumption policy by providing more accurate predictions of likely effects of such individual-level diet changes in a Swedish context.

1.2. Research questions and objectives

The paper represents an in-depth analysis of average Swedish household consumption behavior within the current globalized market framework, linking consumption patterns to their respective environmental load. It thus asks the question: "what are the secondary environmental rebound effects of an average Swedish consumer's shift to a vegetarian diet?" In particular, the study investigates (i) whether vegetarian diets are indeed more environmentally sustainable than the current average Swedish diet; (ii) whether a model vegetarian diet involves significantly different household expenditures; (iii) where Swedish consumers would be most likely to re-spend the potential saved expenditure; (iv) whether important rebound effects negate the initial positive environmental effects of the dietary change; (v) whether this rebound effect is different between low- and high-income households; and (vi) what the most environmentally benign re-spending behavior would look like.

1.3. Research hypotheses

A review of the state of the art, presented in more detail in Section 2.4, allows us to derive the following research hypotheses:

(H1) A shift to a vegetarian lifestyle would, ceteris paribus, incur *lower levels* of energy use and greenhouse gas emissions than the current Swedish omnivore lifestyle;

(H2) A vegetarian diet is on balance *cheaper* than a non-vegetarian diet in Sweden;

(H3) Re-spending of the saved income would lead to *significant rebound effects* in the energy use and greenhouse gas emissions associated with the new consumption pattern;

(H4) The rebound effect will be different between income groups, and is likely to be *larger in low-income households and smaller at higher levels of income*;

and

(H5) The rebound effect could be minimized by focusing the re-spending behavior on the consumption categories that are the *most dematerialized*, for example on services.

1.4. Methods and data

The analysis uses sector-specific Swedish household expenditure data from 2006, which is disaggregated into 117 product categories. This data is linked to expenditure-based environmental intensity indicators (on energy use and CO_2-equivalent emissions) derived from environmentally extended input-output frameworks and life cycle analyses. The shift to the vegetarian diet is based on complete-diet substitutions as proposed by Peters et al. (2007). Following Haque (2005) and Chitnis et al. (2012), likely re-spending behavior is modeled using estimations of the Engel curves of the individual product categories based on four different functional forms, including the Working-Leser and the double semi-log models. This allows both the comparison of different modeling assumptions, as well as a differentiated comparison of rebound effects across household income categories.

1.5. Results and implications

The results indicate that a simple switch to meat- and fishless eating patterns would indeed provide positive savings both in the energy use and the greenhouse gas emissions linked to the consumption behavior of an average Swedish consumer. However, the results also indicate that the model vegetarian diet is indeed cheaper, and that re-spending of these monetary savings would reduce the environmental benefits considerably. Depending on the functional form used, the estimated energy rebound effects range from 95% to 104% of the potential energy savings – that is, leading to no notable change in energy footprint. On the other hand, 49% to 56% of potential GHG emissions savings are not realized due to expenditure on substitution goods. Furthermore, it is apparent that there are large differences in effect between wealthier and poorer households: individuals in the lowest decile are likely to have rebounds of 130% in energy and 88% in CO_2-equivalent emissions, while those in the highest decile only rebound by around 76% and 25% respectively. Convincing consumers to simultaneously switch to organic products, to reduce their workload in order to exchange the monetary savings for more leisure time, or to re-spend the expenditure saved on goods with light environmental footprints such as virtual goods or services might alleviate environmental rebound effects.

1.6. Limitations of the study

The study has to be interpreted within a relatively narrow topical and temporal scope: it studies the theoretical impact of the dietary change and subsequent spending adjustments of a single average Swedish consumer with fixed preferences in the short run and in markets with fixed prices. It thus does not consider long-run market adjustments (regarding prices, demand and supply), changes of preferences, adjustments of work-leisure decisions, and full-scale shifts of land use, and conclusions could vary considerably when such factors are taken into account.

Furthermore, the accuracy of the study's conclusions relies to a great extent on the accuracy of the underlying data; and both environmental intensity data – derived from environmental extended input-output tables and life cycle analyses – and household consumption data – derived from consumer-based recall studies – are prone to simplifications and occasional challenges in representativeness.

Thus, the study's conclusions should be seen in light of the assumptions underlying them, and be treated as indicative results rather than as infallible numeric predictions. Further research on different dietary changes in a variety of regional contexts could contribute to putting these results into perspective and bring even more light in the complex relationship between individual consumption choices and the sustainability of our society.

The remainder of this paper is structured as follows. Chapter 2 reviews why sustainable (food) consumption patterns are of interest for policy-makers, introduces the Swedish context, and motivates the research question by presenting the state of the art and its research gaps. Chapter 3 discusses the theory underlying the analysis of indirect rebound effects and household consumption behavior and summarizes the assumptions made in this study. Chapter 4 focuses on possible methodologies to be used and specifies the choices made in this study. Chapter 5 provides information on the data sources used and their preparation, Chapter 6 specifies the final model used, and Chapter 7 presents its results and sensitivity analyses using several alternative choices. Finally, Chapter 8 draws some conclusions, highlights the study's limitations, and provides ideas for future work.

2. Sustainable Diets: Policy Goals and Research Gaps

An individual's switch to a less meat-heavy or even vegetarian diet can have many interrelated motivations, ranging from ethical and religious reasons to health considerations (Barr & Chapman 2002). The encouragement of such consumption pattern changes has increasingly been considered as a policy option due to the purported environmental benefits of such diets, and therefore their contributions toward more sustainable lifestyles (Steinfeld et al. 2006; Fritsche et al. 2009). This chapter first gives an overview of the political rationale to support sustainable consumption behavior, and in particular sustainable food choices; second, it presents the Swedish policy goals in this issue, particularly regarding meat consumption; and finally, it identifies gaps in the literature related to the investigation of microeconomic consequences of such sector-specific behavior which the present study is hoping to address.

2.1. Motivations behind supporting sustainable consumption patterns

Increasing the sustainability of global consumption patterns is one of the great challenges of our generation. The feasibility of long-term per-capita economic growth has been questioned at least since the 1970s, when Meadows et al. (1972) presented their Limits to Growth model. This framework predicts an inevitable collapse of civilization once non-renewable resources are exhausted, unless human activities are quickly shifted toward a more balanced development that respects ecosystem limits (Meadows et al. 1972). The need for action was also recognized politically during the 1992 Earth Summit in Rio de Janeiro, which called for states to "reduce and eliminate unsustainable patterns of production and consumption and promote appropriate demographic policies" (United Nations 1992, p.2; Barber 2005).

In recent years, a growing body of evidence has surfaced which documents and quantifies the multitude of negative environmental and social consequences linked to current consumption choices. One example of such analyses is the Planetary Boundaries framework (Rockström et al. 2009). Other frameworks include the calculations of national or personal environmental footprints (Hammond 2006), which often showcase radical differences in the ecological burden of different lifestyles, depending on regional diversity as well as conscious decisions about how to lead one's life. For instance, in 2010, the WWF's analysis showed that if every human lived a lifestyle similar to the average resident of the USA or the United Arab Emirates, a bio-capacity (as defined by land required for food production, resources needed for consumption goods and forests and oceans available for carbon sequestration) equivalent to 4.5 Earths would be necessary to sustain humanity. Alternatively, if everybody adopted an average Indian lifestyle, less than half of the planet's bio-capacity would be required (Pollard 2010). Thus, lifestyle choices are clearly of key importance in moving a growing world population towards a safe and sustainable future.

2.2. Sustainable food consumption – an effective policy option?

Improving the sustainability of food choices in particular has received increased attention both in media and policy recommendations over the last years. It is of specific importance since, as Annika Carlsson-Kanyama describes: "food is not easily dematerialised and cannot be substituted for services, commonly proposed as ways to lessen the environmental impacts from consumption of other products than food (OECD, 1996). A change in diet is therefore one of the most important proposals for obtaining sustainable lifestyles in the developed countries" (1998, p. 278).

As with the more general concept of sustainable lifestyles, there has been a rich discussion on how to best define the ideal of a 'sustainable diet'. In 2010, a collaborative effort organized by the FAO arrived at the following common scientific position: "Sustainable diets are those diets with low environmental impacts which contribute to food and nutrition security and to healthy life for present and future generations. Sustainable diets are protective and respectful of biodiversity and ecosystems, culturally acceptable, accessible, economically fair and affordable; nutritionally adequate, safe and healthy; while optimizing natural and human resources" (FAO/Bioversity 2012, p. 83). However, such a broad range of criteria is met with limits in data availability and the quantifiability of certain estimators. Thus, in practice, diets are often decomposed into their components and analyzed with respect to a limited amount of indicators.

Initial investigations in the environmental load of different consumption categories have identified a surprisingly large impact of food purchases, which contribute more than their corresponding share in expenditure to the household's environmental footprint (Goedkoop et al. 2002; Berners-Lee et al. 2012). In Cardiff, for example, almost one quarter of the region's ecological footprint can be attributed to food and drink expenditures according to the analysis carried out by Collins and Fairchild (2007). Furthermore, studies have found important differences in the effect of nutritionally comparable diets in a range of environmental impact factors ranging from land use to effects on biodiversity, greenhouse gas emissions, eco-toxicity, human toxicity, eutrophication and acidification (Hertwich & Katzmayr 2003). Thus, the choice of more sustainable diets is often seen as a relatively effective strategy to minimize private households' environmental loads.

An overview of the current research provides strong evidence that shifting food consumption patterns toward a less meat-heavy diet can have potential positive effects on individual carbon balances and other environmental impacts such as land, water, fossil fuel and chemical input use (Goodland 1997; Dutilh & Kramer 2000; Tukker et al. 2011; Carlsson-Kanyama 1998; Carlsson-Kanyama et al. 2005). Studies have shown, for instance, that the livestock sector contributes as much as 18% of global anthropogenic greenhouse gas emissions (Steinfeld et al. 2006), that animal-based ingredients have ten times higher energy requirements than plant-based ones (Dutilh & Kramer 2000), and that the difference between an affluent, meat-heavy diet and a vegetarian one ranges between a threefold and eightfold higher need for land (Gerbens-Leenes et al. 2002). A number of authors thus concur with McMichael et al. (2007) that a 'contraction and convergence' strategy in meat consumption is required to curb the agricultural sector's environmental impacts.

On the other hand, comparisons of effective consumption pattern changes also show that improvements in the food category are often dwarfed by the potential in other sectors such as housing and appliances. In the EU 27, for instance, the best-case scenario of a shift from meat to vegetable consumption is predicted to save 47 million tons CO_2-equivalent emissions, whereas efficiency improvements in construction, housing appliances and heating systems would result in a total GHG reduction of 700 million tons CO_2-equivalent emissions (Fritsche et al. 2009). It is thus interesting in a first step to evaluate the absolute effect of a sustainable diet change using the present dataset.

2.3. Sweden's interest in sustainable diets

The case of Sweden is of particular interest due to its political and societal focus on sustainability in all areas of life. In Swedish government, environmental policy integration – the mainstreaming of environmental aspects in all areas of policy-making – has been implemented via sectoral responsibility (Engström 2004). In 1998, this measure made 24 public authorities responsible for their sector's ecological sustainability, and specified that this responsibility should include "identifying the role of the sector and how the activities in the sector influence ecologically sustainable development, setting out goals for the sector and encouraging the attainment of these goals" (Engström 2004, p. 9). Thus, the Swedish National Food Agency (Livsmedelverket) has shown extensive interest in the sustainability of Swedish diets and in measures to decrease dietary CO_2 footprints, and report on their findings in publications such as "How small can the climate impact of food consumption be in the year 2050?" (Hjerpe et al. 2013). This report was written in collaboration with the National Agricultural Agency (Jordbruksverket) and the Environmental Agency (Naturvårdsverket) and concluded that a combination of using renewable energies, improving the efficiency of production and distribution, and a change of diets toward more seasonal and vegetarian food could reduce Sweden's food footprint by as much as 30%. Other reports investigate different dietary habits and conclude that switching to an ovo-lacto-vegetarian diet could cut dietary CO_2 emissions in half for Swedish consumers (Clarin & Johansson 2009). As a consequence, in 2009 the National Food Agency submitted a proposal on 'environmentally effective food choices' to the EU in which it advocated for reducing meat consumption, stating that "to eat less meat, and to choose what you eat with care is […] the most effective environmental choice you can make" (Livsmedelverket 2009). In 2013, the Agricultural Agency even called for the implementation of a meat tax to create more sustainable consumption habits (Lööv et al. 2013), and such a step has also been investigated scientifically (Säll & Gren 2012). The topic of meat consumption is thus clearly of high political relevance in Sweden.

2.4. Consumption and rebound effects: State of the art and research gaps

Importantly, though, many of the dietary footprint analyses reviewed above only consider the sector-specific savings derived from changing from certain foods to others. However, multiple inquiries have shown that obtaining nutrients from plants is cheaper than obtaining them from meat or dairy products, that vegetarian diets reduce food costs on average (Lusk & Norwood 2009), and that diet costs increase with the amount of meat consumed (Drewnowski et al. 2004). Yet, the same analysis also showed increased diet costs with increased fruit and vegetable consumption, whereas fats, sweets and carbohydrates were significantly cheaper (Drewnowski et al. 2004). It is thus unclear from previous research whether new (vegetarian) spending levels would be higher or lower, but they are unlikely to stay exactly the same.

With different spending levels, however, sector-specific analyses will ignore the consequences due to a redistribution of expenditures, holding income – and thus, total consumption levels – constant (Alfredsson 2004). This is typically called an *indirect rebound effect* and is operationalized as the percentage of potential savings in particular environmental impacts of consumption that were *not* realized due to consumers' rebound (in particular, re-spending) behavior (Druckman et al. 2011). Its definition and theoretical underpinnings will be further discussed in Chapter 3. Though the literature on the indirect rebound effects of consumption pattern changes is only in its infancy, a number of first estimations concerning a variety of potential lifestyle shifts have been carried out.

One of the first studies in this area, Chalkley et al. (2001), looks at the likely re-spending behavior resulting from the use of more energy-efficient household appliances in the United Kingdom. They find energy use rebound effects between 22% and 27%, and thus conclude by cautioning that re-spending behavior should be considered as an important factor in sustainability-enhancing policies. The findings by

Chitnis et al. (2012) also focus on energy efficiency improvements in UK dwellings, are measured in greenhouse gas emissions, and show a range of 5 – 15% rebound; mainly because the goods and service categories that re-spending occurred in are less greenhouse gas (GHG)-intensive than energy consumption itself.

In comparison, measures that investigate consumption pattern changes in relatively less energy- or GHG-intensive sectors – including dietary decisions – find much larger rebound effects. This makes intuitive sense if the assumed re-spending of the avoided expenditure occurs in categories with higher environmental loads such as household fuels and personal transportation. Druckman et al. (2011) focus on UK household carbon footprints. Here, a sustainable consumption change in the food sector (eliminating food waste) is associated with a rebound effect of 59%, the highest of four simulated scenarios. Alfredsson (2004)'s model of a 'green', more plant-based diet even shows that though the new diet is lower in energy use and CO_2 emissions, re-spending the savings leads to a backfire effect where both energy and CO_2 footprints increase by 2% compared to the original values. The greatest re-spending categories are, in order of magnitude, travel, recreation, food, clothes, and housing. Carlsson-Kanyama et al. (2005), on the other hand, find even greater environmental savings than anticipated from a shift to an energy-efficient diet; but this is mainly due to their assumption that 'green' consumers will purchase organic products and incur larger costs than in the original scenario.

Finally, Lenzen and Dey (2002) find that a switch to a more healthy diet, represented by the recommended dietary intake (RDI), would lead to net savings in CO_2 emissions of around 20%. Yet, they also identify rebound effects of around 50%. On the other hand, the energy footprint shows net increases in energy use of 4 – 7% compared to the original consumption pattern, representing a rebound effect of 111 – 123%. This analysis also estimates re-spending and rebound effects for three consumer categories – the lowest, middle and highest income quintiles – separately, allowing for a more rich evaluation of the effectiveness of such changes. The backfire effect is largest for the lowest income quintile, showcasing their tendency to re-spend the saved expenditure in relatively energy-intensive sectors. This conclusion is supported by Murray (2013), who confirms in his analysis of 'green' consumption decisions (such as reduced vehicle or electricity use) that "in both the conservation and efficiency models the total rebound effect, and both the direct and indirect effects individually, were inversely related to household income level" (Murray 2013, p. 247).

This review points to several previous insights into the role of food consumption patterns and environmental impacts when seen from a rebound framework. First, the definition of 'sustainable diets' is extremely broad and studies have utilized a variety of measures to transform the ideational concept into concrete change scenarios. This makes a direct comparison of results difficult. Overall, it seems that dietary changes, unless accompanied by substantial increases in food expenditure as in Carlsson-Kanyama (2005)'s example, have a comparatively small net impact on energy use and greenhouse gas emissions and high rebound effects. This is mainly due to their relatively low energy intensity compared to other consumption categories. However, as Lenzen and Dey (2002) and Murray (2013) show, there can be interesting differences in effect depending on household income.

In general, there only exists a limited amount of work in this area of research so far. As far as I could identify during my literature review, no previous study has studied the switch towards a vegetarian diet in a rebound framework. Yet, ignoring such substitution behavior might seriously overstate the environmental benefits attached to isolated consumption behavior changes, leading to faulty conclusions that sector-specific changes are sufficient to lead society toward a more sustainable lifestyle. This study can thus inform public policy decisions by contributing to more accurate predictions of the effects of individual-level consumption choices such as the switch to a less meat-heavy diet suggested by the Swedish National Food Agency, and help to identify possible strategies to mitigate undesirable consequences of expenditure adjustments.

3. Theoretical Framework

The rebound concept originates from a long scholarly tradition, though it has only recently been applied to consumer behavior. This chapter reviews the concept's delineation and the definition used for an application to consumer-led abatement actions. It then turns to microeconomic theory and the axioms related to the investigation of household consumption demand systems. Thereafter, it discusses the implications and limitations behind the neo-classical assumption of fixed preferences in the present context. Finally, it introduces the theory behind the use of Engel curves and marginal expenditure shares.

3.1. The rebound effect

Scholarly attention to rebound effects initially grew out of the paradox that global per capita energy consumption kept increasing in spite of important technological advancements in energy efficiency (Sorrell 2007). The concept was therefore concerned with the gap between the potential (PES) and the actual energy savings (AES) stemming from an increase in energy efficiency of a given sector. As Thomas and Azevedo (2013) summarize, within the energy policy and engineering literature the mathematical formula for the rebound effect (R) is often given as:

$$R = 1 - \frac{AES}{PES}.$$

It thus describes what percentage of the potential energy savings was not realized due to interaction effects related to the efficiency improvements (Berkhout et al. 2000).

This section will first provide a broad overview of the different types of rebound effects currently under investigation, before turning to a precise definition of the rebound effect studied in this paper.

3.1.1. Direct, indirect and economy-wide energy rebound effects

In his seminal review on rebound effects, Sorrell (2007) stresses the importance of defining the rebound effect one is estimating in relation to a particular time frame (short, medium, or long term) as well as its system boundaries (such as household, firm, sector, or national economy). Following these requirements, interaction effects can be decomposed further into direct, indirect, and economy-wide components, as done by Sorrell (2007), Thomas and Azevedo (2013), and Hertwich (2005).

Direct energy rebound effects describe an increased use of the appliance or activity in question at the household level due to the decrease in price associated with energy efficiency improvements. For example, in a car that requires 5% less fuel per kilometer driven, driving one additional kilometer will become 5% cheaper compared to the previous status quo. Thus, it is likely that consumers will re-use some of the energy savings embedded in the increase in fuel efficiency by driving more. Sorrell (2007, p. 4) here distinguishes the substitution effect, "whereby consumption of the (cheaper) energy service substitutes for the consumption of other goods and services while maintaining a constant level of 'utility', or consumer satisfaction", and the income effect, "whereby the increase in real income achieved by the energy efficiency improvement allows a higher level of utility to be achieved by increasing consumption of all goods and services, including the energy service". To be precise, however, in most of the literature the increased consumption in other sectors than the one where efficiency improvements occur is considered an indirect rebound effect as described below.

Indirect energy rebound effects, in contrast, describe the consequences of re-spending some of the energy costs saved due to the efficiency investments on other energy-intensive goods. For instance, saving 5% of one's fuel costs over an entire year might sum to a significant amount of money that could then be re-spent on, for example, a weekend getaway to a new city. If that destination is reached by airplane, the additional energy use might even outweigh the yearlong energy savings reached by the fuel-efficient vehicle. In that case, one speaks of *backfire* effects – ironically, the individual's overall energy use balance would have been lower in the status quo than in the situation stemming from efforts to decrease that individual's energy use.

Finally, one has to consider *economy-wide* energy rebound effects. These occur when demand and supply interactions lead to macroeconomic shifts stemming from the original efficiency increase. For example, the increase in fuel-efficiency might on balance lead to a decreased demand for fossil fuels, which in turn might lower prices – which would in a further step give an incentive for higher fuel consumption both in driving and in other consumption categories. Other macroeconomic effects, according to Thomas and Azevedo (2013, p. 200), include changes in economic structure, economic competitiveness, investment and disinvestment, and labor market changes. Many analyses also add direct and indirect rebound effects to the macroeconomic changes such that the economy-wide rebound effect is defined as grand sum of all possible consequences resulting from an initial shift in energy efficiency.

Figure 1 provides a visual representation of the different effects under consideration.

<table>
<tr><td colspan="4">Rebound Effects</td></tr>
<tr><td colspan="2">Direct Effects</td><td>Indirect Effects</td><td rowspan="2">Macroeconomic shifts</td></tr>
<tr><td>Substitution</td><td>Income</td><td>Effects</td></tr>
<tr><td colspan="4">Economy-wide Effects</td></tr>
</table>

Figure 1: Overview of rebound effect definitions

Chitnis et al. (2012) introduce a further delineation in the scholarly analysis of environmental rebound effects by differentiating between the consumption and production perspectives. The consumption perspective includes emissions that arise overseas when they are embedded in products consumed domestically, but excludes emissions that arise within a country linked to products that are subsequently exported. The production perspective, on the other hand, includes all emissions that arise from the production process that occurs within a given country, but excludes 'imported' emissions.

3.1.2. Definition of indirect environmental rebound effects

While the concept of rebound effects originated in the energy efficiency literature, it has since been applied to a wider range of options resulting in prima facie more sustainable consumption patterns. In a broader application of the concept, the rebound effect (R) can be then expressed in the following manner (Druckman et al. 2011):

$$R = 1 - \frac{Actual\ savings}{Potential\ savings} = \frac{Potential\ savings - Actual\ savings}{Potential\ savings}\ .$$

This effect can be applied to different impacts of particular behavior, such as energy use related to a particular individual's consumption pattern or the CO_2-equivalent emissions. It then represents the percentage of potential savings in these impacts that were *not* realized due to consumers' rebound (in particular, re-spending) behavior. However, it does not necessarily have to be positive; if the new behavior incurs higher expenditures in the particular categories than previously, a 'negative rebound effect' could occur where savings are even larger than anticipated due to reduced expenditure in other categories.

When applying the rebound framework to the simulation of consumer-led shifts to alternative lifestyles – which Druckman et al. (2011) call 'abatement actions' – there will be no direct rebound effects, since consumers consciously chose to reduce the consumption of the particular good. Depending on the scope of analysis, the rebound is therefore reduced to indirect and possibly economy-wide effects.

In mathematical terms, the rebound effect can thus be written as follows:

$$Rebound = \frac{\Delta E_P - \Delta E_A}{\Delta E_P} \tag{1}$$

in which

$$\Delta E_P = E^{Original} - E^{First\ Round};$$

and

$$\Delta E_A = E^{Original} - E^{Second\ Round}.$$

Here, 'Original' relates to the original (observed) behavior (and $E^{Original}$ to the energy use or, respectively, greenhouse gas emissions associated with that behavior); 'First Round' represents the situation where exogenous behavior changes were made, but no re-spending has occurred; and 'Second Round' signifies the sum of exogenous and endogenous changes in consumption, where the endogenous changes were derived using household consumer theory.

Thus, the difference between the original and the first round emissions are the intended – or potential – effects of a more sustainable behavior; whereas the difference between the original and second round emissions is our best estimate of the "true" emissions savings according to economic theory. In Figure 2, for instance, the actual savings only constitute 60% of the potential savings. We therefore conclude that there was a rebound effect of 0.40, or 40%. Backfire effects occur when rebound effects rise over 100%. On the other hand, if the actual savings are larger than 100% of the potential savings, we observe virtuous cycle effects where one sustainable behavior reinforces the overall sustainability of the household's lifestyle.

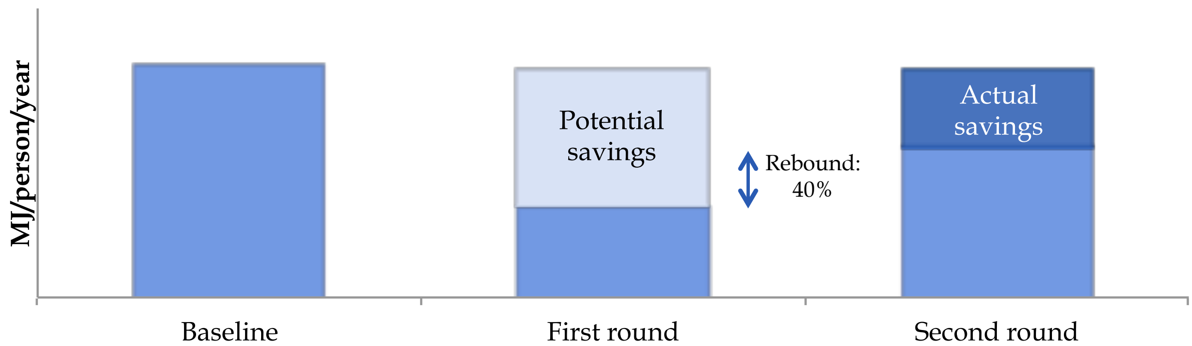

Figure 2: Graphical representation of energy use rebound effect

3.2. Household consumption behavior in a neo-classical framework

Having defined the measure of interest for this study, it is necessary to also specify the theoretical framework that will be used to forecast the 'actual' consumption behavior. In particular, we are interested in the most accurate way to predict the re-spending behavior at the core of the indirect rebound effect.

In a first step, this section states the basic axioms of microeconomic household consumption analysis. It then focuses on the implications of utility maximization following a behavior change in a framework of fixed preferences. Thereafter, it reviews household consumption demand systems and the use of Engel curves to model the relationship between income and consumption. It also defines the interrelated concepts of marginal expenditure shares and income elasticities. Section 3.3 finally summarizes some implications of using the neo-classical framework for the present study.

3.2.1. Neo-classical axioms of household consumption behavior

The investigation of household consumption behavior in general, and rebound effects in particular, can be situated within the framework of neo-classical economic principles that explain and predict the behavior of individual economic actors. This framework is characterized by a set of axioms and assumptions that are specified below. For further reading and an interesting discussion of these principles in a rebound framework, the reader is referred to Blundell (1988) and Berkhout et al. (2000).

First, according to the *rationality* principle, the economic agent obeys preferences that are *transitive* (when good A > [is preferred to] B and B > C, it follows that A > C) and *insatiable* (more is always preferred to less). Furthermore, the economic agent is assumed to follow *optimizing* behavior; in particular, households are presumed to be *utility maximizers*. This utility can come from various sources, but in neo-classical economic theory the most important sources of utility under consideration are the consumption of goods and services as well as the enjoyment of leisure time.

The second leading principle is that of *certainty* and *complete information*. In the neo-classical framework, economic agents can make decisions based on all knowledge about present and future conditions relevant to behave rationally. They thus do not have to consider events that might happen with a non-zero non-unit probability such as uncertain price changes or changes in product availability.

A third important axiom when specifying consumption decisions is that of inter-temporal and inter-category *weak separability* of the utility function. This implies that economic agents make their optimizing choices in stages, and that all information related to a previous stage is irrelevant once decisions have been made. For instance, once all savings decisions for each period are made, price changes in future periods have no independent effect on current period allocations of income. Similarly, the separability of goods and leisure is crucial when consumption and labor supply decisions are modeled separately.

Fourth, individual consumers and households are said to be *price-takers* and their consumption decisions in isolation are not considered to have enough power to significantly affect prices in the wider market.

Finally, economic agents are presumed to constantly adjust to new circumstances by moving to their new optimum state derived from their utility-maximizing calculations. These adjustments occur smoothly and *adjustment costs* in transitioning from one state to another are negligible in a neo-classical framework.

When applying these axioms to an investigation of isolated household consumption decisions, as the present study intends to do, it follows that a representative individual agent undergoes a three-step process in her consumption decisions: first, she will decide on her optimal allocation of labor and leisure time and decide on the optimal amount of working hours, from which she will derive her income; second, she will make her savings decision based on her intertemporal consumption smoothing preferences; and third, she will spend the entire rest of her disposable income on an assortment of goods and services according to their marginal contribution to her utility.

3.2.2. Behavior change, preferences, and utility maximization

With this basic framework in place, we next need to consider how these neo-classical assumptions relate to the modeling of the initial behavior change, that is the shift from $E^{Original}$ to $E^{First\ Round}$, as well as subsequent adjustments.

Vegetarians have been proven to be statistically different from average consumers in a variety of socio-economic factors, as well as their values and purchasing choices of non-meat items (Barr & Chapman 2002). This raises the question whether we aim to simulate a total shift of the average consumer to an average vegetarian mindset, or simply impose the main defining restriction – the absence of meat and fish products in the diet – and leave all other preferences constant.

The first scenario would imply that consumers' preferences and perceived utility could change considerably, and would lead to important questions regarding their rebound behavior – for example, if they derive the same utility from a cheaper vegetarian diet as from the previous omnivore one, they could either decide to stay at the same income and increase their utility, or to stay at the same level of utility and decrease their working hours to adjust their income. Furthermore, since their re-spending behavior depends on their perceived marginal utility of different consumption goods, the new vegetarian re-spending behavior might be significantly different from an average consumer's. In the second scenario, in contrast, we would expect a total re-spending of income saved in order to recoup as much as possible of the utility lost by foregoing meat. Similarly, this income would be re-spent according to the observable preferences derived from current consumption behavior.

The question requires the consideration of broad-ranging taste and preference changes among consumers, an area that has only been given limited thought within a neo-classical economic framework, within which preferences are generally seen as stable (Kallbekken et al. 2008). Von Weizsäcker (1971) differentiates between purely exogenous influences on taste, which he says will not alter the efficiency or Pareto optimality properties of microeconomic models, and endogenous factors, which may require changing the conceptual framework of the theory itself. His analysis however concludes that consumption pattern changes, if they do exist, are incremental over time and would mainly influence long-run demand curves. Pollak (1978) comes to the same conclusions in his treatment of habit formation. Coming from an empirical perspective, Burton and Young (1992) estimate static and dynamic demand systems for meat and fish and conclude that systematic, exogenous demand shifters related to consumers' changing tastes and preferences in response to external information can indeed be identified. However, they are unable to pinpoint the exact source of these shifters. Also, their analysis spans 30 years, from 1960 to 1990. Yet, it seems plausible that taste changes can indeed be induced by external factors, such as for instance government information campaigns or increased awareness about issues related to health or animal welfare.

On the other hand, Lusk and Norwood's (2009) estimation of the willingness to pay for meat consumption concludes that for current American omnivore consumers, meat is the most valuable food group in their consumption basket. They do recognize that "it could be that the demand and utility parameters used to calculate the value of meat consumption would change once people became accustomed to the vegetarian lifestyle." However, they caution that "these are long-term considerations, but persuading individuals to give up meat requires short-term adjustments. The fact that meat is the most valued food source indicates that it would require considerable persuasion for the average American to forgo meat" (Lusk & Norwood 2009, p. 121–122). Thus, there seems to be a strong case for combining a short-term analysis of effects with a framework that assumes constant tastes and preferences, and in which the shift to vegetarianism stems from a government policy or a persuasion effort which did not achieve to change underlying tastes and preferences of the average consumer in-depth.

3.2.3. Household consumption demand systems

These tastes in turn are most often observed as revealed preferences through households' expenditure choices. They can be further described in the context of household consumption demand systems, which will be explained in more detail below.

The ordinary demand function

It has been well established in microeconomic consumer behavior theory that the constrained utility maximization problem of the individual consumer leads to the following ordinary demand function for individual goods:

$$x_i = f(p_1, p_2, \dots, p_i, \dots, p_n, Y) \tag{2}$$

in which x_i = the quantity of the i^{th} good or service consumed;
p_i = the price of the i^{th} good or service;
and Y = income.

Following the axioms laid out in the previous section, ordinary household demand systems are then characterized by the following restrictions:

The budget restriction: $\sum_i p_i x_i(P, Y) = Y$.
The requirement for homogeneity of degree zero in income and prices: $Y \frac{\partial x_i}{\partial Y} + \sum_j p_j \frac{\partial x_i}{\partial p_j} = 0$.
And the symmetry of the substitution effect described by the Slutsky relations:

$$\frac{\partial x_i}{\partial p_j} + x_j \frac{\partial x_i}{\partial Y} = \frac{\partial x_j}{\partial p_i} + x_i \frac{\partial x_j}{\partial Y} .$$

For further insights on household demand systems, the reader is referred to Haque (2005).

In order to correctly parametrize this system of equations, it is common to reduce the dimensionality of the problem. In the context of re-spending rebound effects, it is of particular interest to regard the case when prices are kept constant and the focus of analysis is the response of relative expenditures to changes in income.

Engel curves as measures of relative expenditures

As Haque (2005) details in his in-depth investigation of consumption behavior in relation to economic development, modeling the consequence of changing budget constraints is commonly achieved by estimating Engel curves.

Engel curves describe the relationship between the expenditure on a particular commodity and total income. Thus, they can be represented in general as

$$x_i = f(Y) \tag{3}$$

in which x_i = expenditure on commodity *i;*
and Y = income, represented by total expenditure.

Note that following the axioms of utility maximization and weak separability, the theoretical assumptions of keeping prices fixed and analyzing one isolated time period allow us to redefine some of the variables of the original demand function (2). With fixed prices, the amount of commodity consumed can be equivalently measured by the expenditure on the particular commodity. Furthermore, when assuming that inter-temporal utility maximization decisions, such as deciding on the optimal spending ratio, have been made prior to consumption decisions in the current period, and that agents maximize utility by exhausting their budget constraint, one can represent disposable income in the current period by total expenditure. This static analysis thus does not allow for advanced dynamic predictions and relies on data collected within a short amount of time and in a region where prices are reasonably similar (Haque 2005).

As Haque explains, one important theoretical assumption underlying Engel curve analyses using household expenditure survey data is that on average, "the differences in consumption patterns between high and low income households can be ascribed to their differences in current income (total expenditure)" (2005, p. 7). Thus, the analysis is more informative the more homogenous the group of households is in socio-economic characteristics that could have a significant effect on preferences, such as educational and cultural background, occupation, age and sex composition of households, and in particular the family size of the households.

Cramer (1973, p. 147) points out that household size is strongly correlated with total household expenditure. Therefore, controlling for it in a regression framework can lead to large standard error terms. It is thus common to re-specify equation (3) in terms of per capita expenditures, such as:

$$\frac{x_{ij}}{s_j} = f(\frac{Y_j}{s_j}) \tag{4}$$

in which x_i = expenditure on commodity *i* by household *j*;
s_j = size of household *j*;
and Y_j = income of household *j*, represented by total expenditure.

When measuring household size by counting the number of individuals, however, a randomly selected sample of $\frac{Y_j}{s_j}$ would be negatively correlated with the number of children in the household. Haque (2005, p. 192) argues that is thus more consistent to equalize expenditure by measuring s_j in adult-equivalent consumption units. Section 4.3.3 goes into more detail on the econometric specification of household size and the calculation of equivalized expenditure.

Marginal expenditure shares and income elasticities

Using Engel curves, we can analyze household spending patterns, particularly their behavior in relation to an income increase, through the use of two connected concepts which can be used separately or in tandem: (i) marginal expenditure shares and (ii) income elasticities.

(i) The simpler method is to use the *marginal expenditure share* (MES), also known as the *marginal propensity to spend*, which is simply defined as the partial derivative of the expenditure on a particular good or service to total expenditure. It therefore represents the slope of the Engel curve, or mathematically (in keeping the Engel specification of (4)):

$$MES_i = f'\left(\frac{Y_j}{s_j}\right) = \frac{\partial\left(\frac{x_{ij}}{s_j}\right)}{\partial\left(\frac{Y_j}{s_j}\right)}. \tag{5}$$

These marginal values are very informative when investigating the probable re-spending distribution of one additional unit of income. They are thus used by both Alfredsson (2002) and Murray (2013) in their rebound studies. As Alfredsson argues, using marginal expenditure shares "has good theoretical/empirical justification, since it ensures that the loss of utility from adopting a change in consumption pattern is minimised whenever a consumer's utility function is of the Stone Geary type" (2004, p. 515).

(ii) However, since the MES represent the ratio of absolute and not of relative change, and are a more specialized measure, many authors prefer to define the responsiveness of consumers to income changes by using *income elasticities*, the logarithmic partial derivatives. They are defined as follows:

$$\eta_i = \frac{\partial \log\left(\frac{x_{ij}}{s_j}\right)}{\partial \log\left(\frac{Y_j}{s_j}\right)} = \frac{\left(\frac{Y_j}{s_j}\right)}{\left(\frac{x_{ij}}{s_j}\right)} * \frac{\partial\left(\frac{x_{ij}}{s_j}\right)}{\partial\left(\frac{Y_j}{s_j}\right)} = \frac{\left(\frac{Y_j}{s_j}\right)}{\left(\frac{x_{ij}}{s_j}\right)} * MES_i\,. \tag{6}$$

From (6), we can see that there is a direct relationship between the concepts of marginal expenditure share and income elasticities. Thus, it is possible to use one measure to arrive at the other, or vice versa. In addition, with the exception of a linear specification of the Engel curve, the definition shows that both marginal expenditure shares and income elasticities will depend on total income and may be significantly different between different income deciles (Chitnis, Sorrell, Druckman & Firth 2012; Haque 2005). As Murray (2013) argues, this is particularly interesting for a rebound analysis, where one might expect a large variation in the size of the environmental rebound effect depending on whether re-spending occurs on necessities or on luxury items, which have been shown to have lower greenhouse gas intensities on average.

Finally, we can also express the Engel aggregation condition using both concepts. This theoretical restriction is derived from utility theory and stipulates that all marginal changes in expenditure must equal the total change in income. Most commonly, the Engel aggregation is expressed by defining the budget share of the i^{th} good as

$$W_i = \frac{\left(\frac{x_{ij}}{s_j}\right)}{\left(\frac{Y_j}{s_j}\right)}.$$

We then arrive at the following restriction, which has an equivalent expression using marginal expenditure shares:

$$\sum\nolimits_i W_i \eta_i = 1 \stackrel{\text{def}}{=} \sum\nolimits_i M\mathrm{ES_i} = 1\,. \tag{7}$$

When estimating Engel curves, it is important that this theoretical restriction is respected (Haque 2005, p. 6).

3.3. Application of theoretical framework

This overview has shown that there are a number of theoretical choices to make that need to be laid out transparently because they will influence the interpretation of the results significantly. This section will restate these choices with regard to the research question at hand, and will discuss implications deriving from them.

First, the use of the *neo-classical household consumption framework* underscores the drawbacks of any conclusions on environmental improvements of consumption choices that do not take further adaptive behavior into consideration. An agent that would refrain from re-spending the savings incurred by changed expenditures would behave suboptimally under the neo-classical assumptions and could *increase her utility* significantly by exhausting her budget constraint in any given period.

Second, it makes apparent that the choice between short- and long-term adjustment processes will influence both the analytical choices and the conclusions drawn from these types of study significantly. This study limits its analysis to *short-term adjustment behavior*. This means that we automatically choose to treat employment and savings choices as exogenous and constant, in line with the weak separability axioms. Thus, our results might be different from those that take a more long-term perspective; here, one could expect consumers to reassess, for instance, their work-leisure trade-off and choose to reduce their working hours in order to keep consumption – after the dietary switch – constant and to match their required income to the new decreased expenditure. However, such an integration of labor and consumption decisions goes beyond the scope of this study.

The analysis is also *limited in scale*, focusing on the impact of an individual consumer's behavior change. It therefore does not consider macroeconomic changes resulting from wider-ranging adjustments in supply, demand and prices. These – undoubtedly important – considerations would need to be investigated further in a latter step using different data and methods from the ones utilized here.

Furthermore, choosing a short-term perspective also implies the assumption that *preferences remain constant*, and that the only change considered is the switch to a nutritionally sound vegetarian diet. This provides another rationale why consumers could be relied on to re-spend the entirety of the saved income: if forgoing meat leads to a decline in their utility, it would be most likely that economic agents try to recoup that utility by consuming more other goods. Constant preferences would equally imply the use of marginal expenditure shares as the best-possible approximation of the current marginal added utility of the different consumption categories, whereas the assumption of the existence of preference changes opens up a plethora of possible rebound behaviors, for example purposefully 'green' re-spending on organic products or services. In order to reach richer policy insights, we relax the assumption of constant preferences and test several of these alternatives in the sensitivity analysis.

4. Methodology

Assessing ecological rebound effects from a microeconomic household consumption framework necessitates an accurate estimation technique of the environmental footprints linked to households' consumption behavior. It furthermore requires an in-depth understanding of both theoretical and empirical factors influencing consumption choices in order to construct theoretically sound and empirically viable simulations with plausible results. This chapter first reviews the state of the art in environmental impact assessments in Section 4.1. Thereafter, it discusses convincing methods to move from the old to the new consumption behavior. Section 4.3 presents an overview of methods to estimate rebound effects and provides insights into household consumption econometrics. Section 4.4 finally specifies the concrete methodological framework used in the current analysis along with a rationale of the choices made.

4.1. Estimating the environmental impacts of consumption choices

In order to attribute environmental impacts accurately to individual households or persons – and thus to reach values for $E^{Original}$, $E^{First\ Round}$ and $E^{Second\ Round}$ – it is necessary to combine household consumption data with the environmental intensity embodied in the products used. In the discipline of industrial ecology, such intensity parameters are usually derived using life cycle analyses (LCA) and environmentally extended input-output (EEIO) tables. Such tables resemble input-output models used in Social Accounting Matrix designs, but are adapted for environmental footprint analyses by adding in all related environmental impacts at every step of the supply chain. Because the intensities are highly context-specific, they need to be calculated on a national, or even more ideally, regional level. This has been done for Sweden (Carlsson-Kanyama et al. 2005; Brännlund et al. 2007), the UK (Druckman et al. 2011), Japan (Mizobuchi 2008), the United States (Weber & Matthews 2008), the Netherlands (Biesiot & Noorman 1999), as well as for 11 other EU countries (Reinders et al. 2003). Notwithstanding the recognition that environmental consequences of human production and consumption patterns are multifaceted and interconnected (Dernini et al. 2013), most of these EEIO tables focus on a select number of indicators for simplicity's sake. Most common is the focus on carbon footprints (Weber & Matthews 2008; Druckman et al. 2011; Brännlund et al. 2007; Mizobuchi 2008) and energy use (Carlsson-Kanyama et al. 2005; Reinders et al. 2003), or a combination of both (Biesiot & Noorman 1999).

When applying these intensity parameters in footprint and rebound studies, there are certain challenges. Many of the present studies (Alfredsson 2004; Brännlund et al. 2007; Lenzen & Dey 2002; Mizobuchi 2008) use relatively old data – stemming from the 1990s – both when analyzing consumption patterns and when calculating environmental intensity indicators. This might lead to imprecise conclusions considering the rapid changes in our global food and trade systems. Furthermore, Carlsson-Kanyama et al. argue that if one adapts environmental data from other countries, as Alfredsson (2004) did when using Dutch data in a Swedish context, "there may be substantial deviations and modeling may give unreliable results" (2005, p. 230). Thus, it is important to pay attention to using country-specific and up-to-date data which corresponds both at both production and consumption level.

4.2. Modeling the switch to a vegetarian diet

Understanding the substitution behavior of consumers switching from non-vegetarian to vegetarian consumption patterns is important both for modeling purposes as well as for adequate policy support. It is a particular challenge since the vegetarian diet is characterized mainly by omission – excluding the consumption of meat and fish products -, and the final composition of that diet is thus subject to large variation (Key et al. 2006).

In one study (Schösler et al. 2012), flexitarians (defined as participants that reported to eat meat less than five times a week and to replace it with other protein options otherwise) were given nine replacement options and were asked to pick the three they were most likely to consume instead of meat. Participants tended towards the selection of fish, eggs and cheese, all animal-based products that might not provide a large advantage over meat considering their environmental impacts. These results showcase the multidimensionality of sustainable consumption patterns.

According to Alcantud et al. (2009), on the other hand, various nutrition studies find that a reduction of meat in the diet leads to an average increase of 25% of other food groups. They also point out that other food groups are significantly different in their energy value (calculated in protein) compared to meat, with most of them exhibiting higher required ratios of weight-to-energy ranging from 2.2 (dairy products) to 8 (fruit) times the amount of food required for the same levels of protein intake.

In their footprint analysis, Collins and Fairchild (2007) replace beef, cooked poultry and fish consumption with cheese; mutton, uncooked poultry and lamb consumption with eggs; and pork, ham, and other meat consumption with cereals. The replacement was seemingly done according to weight, which might not adequately represent the typical consumption pattern of consumers. It also resulted in a nutritionally inadequate diet, with 8 out of 19 nutrients falling under the reference nutrient intake. The results of this rebound scenario should therefore be interpreted with care.

As Peters et al. (2007) argue, the problem with many of these approaches is that they try to isolate dietary substitution steps, rather than appreciating holistic food consumption patterns in complete-diet comparisons. Therefore, modeling a switch to a vegetarian diet by replacing meat gram-for-gram by cheese, for example, seems overly simplistic and removed from reality. Rather, it seems informative to study the actual diets of comparable samples of vegetarians and non-vegetarians and investigate average differences in the consumption of a range of food products.

For the purposes of this study, there exist two studies that provide such comparative results. The first, Larsson (2001), was interested in the consumption patterns of young vegetarians and vegans in Sweden in the 1990s. Compared with omnivores of the same age group (15-year olds), female vegetarians were reported to eat significantly more servings of vegetables per month, though not as much as would be expected of a vegetarian population. Furthermore, they ate less snacks and sweets than the female omnivores. Male self-defined vegetarians, on the other hand, show no significant increases in food consumption compared to their omnivore counterparts, but only consumed animal products, margarine, ice cream, cookies, and fried and barbecued food less often (Larsson 2001). It is thus questionable whether this subpopulation adequately represents the average consumption behavior of any male vegetarian, which would be assumed to replace the protein and calories foregone by cutting out meat with other products.

The second study, Haddad and Tanzman (2003), was carried out in the United States using nationally representative survey data from over 13 000 respondents that was gathered between 1995 and 1998. Vegetarians were shown to eat more legumes, some vegetables, and fruit than non-vegetarians, but to have relatively similar intake levels of dairy products, sugar and sweets. The broadness of the sample, the high level of disaggregation among the food items, and the thoroughness in data collection led Berners-Lee et al. (2012), who studied the greenhouse gas implications of realistic dietary choices, to utilize this study as base for their vegetarian case study. It will also be further discussed in Section 5.1.2.

4.3. Estimating rebound effects

Due to the interdisciplinary interest in environmental rebound scenarios, there exist a number of competing estimation practices. This section reviews the two main schools of thought before moving to the more practical issues of modeling re-spending behavior using marginal expenditure shares derived from Engel curves.

4.3.1. Approaches from energy economy and industrial ecology

Thomas and Azevedo (2013) differentiate between two schools of analyses that attempt to estimate indirect rebound effects. In the first case, *energy economists* generally use Almost Ideal Demand System (AIDS) models to estimate direct and indirect rebound effects of efficiency savings jointly. This is done by calculating income and price elasticities for energy goods and other consumption categories and extrapolating aggregate economy-wide behavioral changes based on these marginal values. Brännlund et al. (2007), for instance, use Swedish time series data on 13 disaggregated consumption categories and their associated emissions in order to calculate rebound effects resulting from technological progress, as well as the adjustments in CO_2-tax necessary to neutralize them. A similar analysis is also carried out by Mizobuchi (2008) in a Japanese context. However, these approaches suffer from a number of shortcomings, including the disregard for time trends, lagged price variables, and other measures of technological change, as well as a very narrow focus on certain types of emissions. In particular, most center around combustion or purchased electricity emissions and do not include supply chain emissions, which are crucial when examining the embodied energy and emission intensity for different consumer goods (Thomas & Azevedo 2013). Furthermore, AIDS models assume linear Engel curves, which might be a simplification of consumer behavior (Banks et al. 1997). For these reasons, as well as the fact that energy economists' analyses are more concerned with efficiency improvements and less concerned with exogenous demand changes, these methods seem unsuited for the present research question.

On the other hand, such changes, as well as household and national environmental footprints, are extensively studied in the *industrial ecology* literature. In this field, estimations of the energy consumption and greenhouse gas emissions embodied in different products are linked to investigations in household spending patterns to estimate the most likely re-spending behavior stemming from an initial switch in expenditures, as well as the environmental impacts connected to it (Thomas & Azevedo 2013). The household spending patterns are mainly derived from the econometric analysis of survey data on household expenditures (Chitnis, Sorrell, Druckman & Jackson 2012). Because these studies assume exogenous demand shifts and constant prices, their focus is exclusively on the re-spending rebound, and thus the income effect of changing consumption behavior, as discussed in Section 3.1.2. This approach is therefore less concerned with estimating income and price elasticities jointly, and rather with the calculation of appropriate income elasticities and the related marginal expenditure share estimates.

Challenges with this approach include arriving at the appropriate marginal expenditure values to model re-spending behavior correctly. Earlier studies simply assume that saved income will be spent in proportion to current spending patterns (Lenzen & Dey 2002; Takase et al. 2008), while others visually identify particular product categories that seem to be consumed more intensively by higher income households and in their simulation shift all re-spending to these categories (Chalkley et al. 2001; Carlsson-Kanyama et al. 2005). However, in recent years the important difference between average and marginal expenditure patterns has been increasingly highlighted (Nässén & Holmberg 2009), and the incorporation of axioms related to microeconomic consumption behavior has grown in significance.

4.3.2. Estimating income elasticities and marginal expenditure shares

The use of income elasticities and marginal expenditure shares has been favored in the literature because of its methodological straightforwardness, the fact that it does not require the imposition of arbitrary restrictions on consumer behavior such as strong separability, and the ability to use a higher degree of disaggregation (Chitnis, Sorrell, Druckman & Jackson 2012; Sorrell 2010). Econometric methods used to derive them largely depend on data availability and the purpose of the study; the two most widely employed approaches are the use of time series consumption data (Chitnis, Sorrell, Druckman & Jackson 2012; Druckman et al. 2011), and alternatively the estimation of Engel curves, their respective marginal expenditure slopes and income elasticities using household expenditure data (Nässén & Holmberg 2009; Alfredsson 2004; Chitnis, Sorrell, Druckman & Firth 2012).

Druckman et al. (2011), for instance, use quarterly time series data on aggregate household expenditure subdivided in 16 categories, and apply a Structural Time Series Model to estimate long-run income elasticities for the UK. The same model (ELESA) was also used in Chitnis et al.'s analysis (Chitnis, Sorrell, Druckman & Jackson 2012).

However, the use of time series data to estimate income elasticities has been criticized to be inaccurate due to the existence of severe multicollinearity between incomes and prices (Haque 2005). Furthermore, in many instances the methods of data collection have changed over the years, making it difficult to compare and aggregate data between survey versions. Additionally, the econometric set-up and the scarce availability of price indices only allows for a limited amount of disaggregation between product categories, and it is thus more challenging to investigate the important differences of alternative goods' energy and greenhouse gas intensities, for example between different food products. It also shows weaknesses in allowing for differences in spending preferences between households of differing income, effects which can differ in size as well as in sign for a particular commodity.

For these reasons, cross-sectional family budget analysis methods are favored when estimating income elasticities separately, and have shown to yield quite reliable estimates (Haque 2005). Alfredsson (2002; 2004), for instance, uses cross-sectional microdata on 1 104 Swedish households to construct a microsimulation model. She specifies a multivariate general linear model that includes demographic and lifestyle characteristics of the individual households into the set of equations that define consumption choices, and derives marginal propensities to spend from this set of equations using linear regression tools. Nässen and Holmberg (2009), on the other hand, use pooled quintile data and calculate marginal expenditure shares for four different household types through least-square estimation, which they then weigh to arrive at average marginal expenditure share estimates. They do not specify the functional form used for their estimations, though it might be assumed that it is also linear. Finally, Chitnis et al. (Chitnis, Sorrell, Druckman & Firth 2012) test two different functional forms in their estimation of Engel curves, the Working-Leser and the double semi-log forms, on UK household cross-section data, and use the flexibility of these functional forms to calculate different income elasticities according to household quintiles. Murray (2013) follows the same procedure to calculate income-dependent marginal budget shares, also using double semi-log and Working-Leser specifications as well as a linear approximation on a disaggregated dataset of 6 957 Australian households' expenditures on 36 consumption categories. He is additionally the first author to acknowledge the shortcomings of previous studies of environmental conservation measures (Alfredsson 2004; Lenzen & Dey 2002; Druckman et al. 2011), whose use of income elasticities allow for re-spending on the precise items their model consumers had originally cut back on. Murray avoids this problem by reallocating the expected marginal budget shares of the conserved items across the remaining categories (2013, p. 243). The next section will go into more detail on the econometric specification of Engel curves used by these two most up-to-date analyses (Chitnis, Sorrell, Druckman & Firth 2012; Murray 2013).

4.3.3. Engel curve specifications and functional forms

There exists a rich body of work in econometrics that attempts to specify the most accurate functional form to model household demand system equations, and in particular the Engel curve (equation 4) from Section 3.2.3. Independent variables that are often taken into consideration as explanatory factors for the consumption of a particular good, in addition to total income, are household size, degree of urbanity, regional location, age of household reference person, and dwelling type (Alfredsson 2002; Murray 2013). The applicability of these variables depend on the degree of disaggregation within the dataset available; in most pooled datasets, it is difficult to control for more than household size. On the other hand, using grouped means also eliminates the problem of high variability within the individual data, alleviates the issue of dealing with outliers because individual households reported inaccurate data, and sidesteps the problem that "inspection of scatter diagrams of expenditures of different commodities by income groups of a number of individual households shows that the dispersion is so large that any Engel function will give an equally poor fit" (Haque 2005, p. 25).

However, we have established that the strong correlation between household size and household expenditure makes it necessary to calculate equivalized expenditure measures before proceeding with econometric calculations. There are a variety of inherent problems of using per capita data, though: since the needs of members of the household vary with age and sex, inter alia, the cost per person of maintaining a certain standard of living also differs. Furthermore, economies of scale may exist when a household can enjoy a higher standard of living than a relatively smaller household with the same level of per capita income (Haque 2005; Cirera & Masset 2010). To counter these difficulties and make households more comparable, it has become common practice to construct consumer unit scales which transform household compositions in a measure of 'effective number of consumers' in a household. The precise procedure to do this differs, but most use either utility or cost functions to estimate the required expenditure for equivalent standards of living of different household compositions. Section 5.1.3 goes into more detail on the consumer unit scale used for the present data set.

When estimating the precise functional form of the Engel curve

$$\frac{x_{ij}}{s_j} = f(\frac{Y_j}{s_j}) \tag{4}$$

in which x_i = expenditure on commodity *i* by household *j*;
s_j = number of consumption units of household *j*;
and Y_j = income of household *j*, represented by total expenditure,

there are several options.

A simple *linear form* can be specified as

$$\frac{x_{ij}}{s_j} = \alpha_i + \beta_i * \left(\frac{Y_j}{s_j}\right) + \varepsilon_{ij} \, . \tag{8}$$

It assumes that the marginal expenditure share of commodity i, MES_i, is simply equal to β_i, and therefore equal for all consumers, independent of their level of income. Thus, consumers are treated as having identical, consistent preferences at constant prices and there is no severe rearranging of preferences according to their relative income class (i.e. low-income consumers have identical preferences as high-income ones). This is a severe restriction of the model that limits its applicability in empirical work, where considerable differences in expenditure patterns between income groups have been observed (Haque 2005; Murray 2013).

We can alleviate some of these concerns by introducing a quadratic total expenditure term in the *quadratic specification*:

$$\frac{x_{ij}}{s_j} = \alpha_i + \beta_i * \left(\frac{Y_j}{s_j}\right) + \gamma_i * \left(\frac{Y_j}{s_j}\right)^2 + \varepsilon_{ij} \,. \tag{9}$$

Now, the marginal expenditure share MES_i depends on the level of income and can be calculated as $MES_i = \beta_i + 2 * \gamma_i * \left(\frac{Y_j}{s_j}\right)$. This specification therefore allows both for different marginal expenditure patterns and for the modeling of different goods, such as essential and luxury goods, as well as for the case that goods are essential goods for some consumers and luxury goods for others. However, this specification is unable to fulfill the Engel aggregation condition exactly, and results therefore might need to be normalized before further use.

In addition to these models, a number of more complex functional forms have been evaluated for their ability to fit the data even more closely. Leading among them are the Working-Leser and the double semi-log.

The *Working-Leser form* relates budget shares to the logarithm of total expenditure. It thus specifies the budget share as $\frac{\frac{x_{ij}}{s_j}}{\frac{Y_j}{s_j}} = x_{ij}/Y_j$ and regresses:

$$\frac{x_{ij}}{Y_j} = \alpha_i + \beta_i * ln\left(\frac{Y_j}{s_j}\right) + \varepsilon_{ij} \,. \tag{10}$$

The functional form of the Engel curve is therefore

$$\frac{x_{ij}}{s_j} = \alpha_i * \left(\frac{Y_j}{s_j}\right) + \beta_i * ln\left(\frac{Y_j}{s_j}\right) * \left(\frac{Y_j}{s_j}\right). \tag{11}$$

And the MES_i is derived as $\alpha_i + \beta_i * [1 + ln\left(\frac{Y_j}{s_j}\right)]$. The Working-Leser form has been considered as preferable because it is consistent with consumer behavior – allowing to model luxuries, necessities, and inferior goods –, allows elasticities to vary with income, and should automatically satisfy the adding-up condition when using OLS regression (Cirera & Masset 2010).

Finally, the model preferred by Haque (2005) in his seminal overview of the estimation of Engel curves is the *double semi-log*. It is specified as follows

$$\frac{x_{ij}}{s_j} = \alpha_i + \beta_i * \left(\frac{Y_j}{s_j}\right) + \gamma_i * ln\left(\frac{Y_j}{s_j}\right) + \varepsilon_{ij}. \tag{12}$$

The MPS_i is derived as $\beta_i + \gamma_i / \left(\frac{Y_j}{s_j}\right)$. Here, similarly, the adding up condition (as well as other economic restrictions) should be automatically satisfied in OLS (Chitnis, Sorrell, Druckman & Firth 2012), and it has proven to fit household expenditure data best thanks to its flexible form (Haque 2005). However, the ultimate choice of the Engel functional form relies on many statistical and economic criteria, and econometricians have to weigh advantages and disadvantages of each specification (Haque 2005).

The question also arises whether the system of Engel equations should be estimated simultaneously (such as within a Seemingly Unrelated Regression Analysis framework), or whether each equation can be estimated individually. The major advantage of the systems approach is described by Haque (2005, p. 47) as that it "takes account of contemporaneous covariance between disturbances in equations for different cross-equation parameter restrictions that arise from constrained utility maximization". However, the procedure provides no added efficiency than the single equation least squares estimation method if all regressors are the same between all equations, which is the case in the present analysis, and if there are no cross-equation parameter restrictions. Since the Engel functions used in this analysis are not directly derived from utility functions, there are no explicit cross-equation restrictions that apply. Therefore, the estimation of each Engel function separately is appropriate (Haque 2005, p. 34).

Finally, one has the choice of using ordinary or weighted least squares to estimate the Engel curves. Haque (2005, p. 39) recommends to use weighted least squares, taking the proportion of the estimated population in each income class as weights, in order to improve the focusing accuracy of the regressions (Shalizi 2014). This might be warranted since the transformation of households – which had been divided into equally sized deciles – into individual consumption units ignores the fact that there are more consumption units in the richer households. Using ordinary least squares might therefore overestimate the importance of lower-income consumption units in the population compared to higher-income ones.

4.4. Method selection

The research question at hand and the data available limit the choice set of possible methodological options and provide important points to take into consideration when selecting our methodology. To reiterate, the present topic of inquiry is "what are the secondary environmental rebound effects of an average Swedish consumer's shift to a vegetarian diet?".

The present analysis is therefore less concerned with exogenous technological efficiency increases of a particular product, but rather with the consequences of ethically or environmentally motivated choices and the related income effects. Since this scenario represents an exogenous demand shift within an unchanged market framework, in which products, prices and technology remain constant, it seems more suited for an analysis in the context of industrial ecology rather than following the methodologies used in energy economics. It thus takes on a consumption perspective (Chitnis, Sorrell, Druckman & Firth 2012) and focuses on *indirect* rebound effects resulting from exogenous changes of consumption patterns, since many of these exogenous changes – such as the switch to a vegetarian diet – explicitly preclude the possibility of re-spending in the same category.

To model the exogenous demand shift – the shift to the vegetarian diet – I decided to follow the lead of Berners-Lee et al. (2012) and use Haddad and Tanzman's (2003) survey since it is the most complete comparison of non-vegetarian and vegetarian consumption patterns available up-to-date. Due to the short-run temporal framework I chose, I judged it unlikely that tastes would have time to change dramatically, and I thus chose to assume constant preferences. I therefore assumed separability between nutritionally essential dietary expenditure and the remaining purchasing behavior, and created a nutritionally sound baseline scenario by only scaling up those items that were consumed significantly more by vegetarians and that could conceivably substitute either the caloric or protein requirements that the lack of meat created. This meant that differences in luxury items such as wine or sweets were excluded from the adaption process. In order to test how far this assumption influenced results, the sensitivity analysis will include a scenario where the entire new consumption pattern of vegetarians is adapted.

Furthermore, since my analysis is concerned with consumption patterns at an individual and household level, it is informative to use household expenditure survey data. These consumption patterns have to be linked to energy and greenhouse gas intensity data, which are ideally consistent in time and place with the consumption data; that is, it requires data that stems from the same country in the same year. Previous work by the Swedish Defense Research Agency (Johansson et al. 2010) has provided very detailed estimates of the environmental intensity of different consumption categories for the year 2006, and I therefore chose to use the most disaggregated household expenditure data for the same year made available by Statistics Sweden, which lists household consumption of 117 goods by income decile. Unfortunately, more disaggregated individual household data is not available for researchers at the Master's level for reasons of confidentiality. Furthermore, Statistics Sweden does not provide price indices or estimates for all 117 categories of goods and services. These limitations in data availability preclude the use of AIDS models or more sophisticated demand system models that take individual household characteristics into account. I therefore concentrated on approximating the most likely re-spending behavior of consumers at that particular point in time, namely the year 2006. This requirement makes the use of Engel curves and the derivation of marginal expenditure shares from them more intuitive than the derivation of income elasticities from time series data, especially since it can be based on the same cross-sectional expenditure data already used to construct baseline consumption and environmental footprint estimates.

After surveying the literature, I decided to use all four of the presented alternatives of the Engel curve estimation and to compare the results both in terms of marginal expenditure shares and in rebound effects in order to additionally provide insights in the sensitivity of rebound studies to the econometric functional form used. In line with the argumentation provided in 4.3.3, I estimated all equations individually using both ordinary and weighted least squares estimation techniques and compared results to prevent a bias due to the selection of the method.

I also chose to follow Murray's (2013) lead in adjusting the marginal expenditure values to preclude re-spending in categories that are explicitly avoided by consumers in their new expenditure patterns. To meet the goal of holding consumers' preferences constant within the restriction of refraining from consuming meat and fish, it seemed most consistent to redistribute the marginal expenditure shares of meat and fish products across the remaining categories in proportion to their relative size within the summation of non-meat and fish products. Furthermore, the sensitivity analysis also includes alternative re-spending scenarios to test whether more persuasive government policies (such as information campaigns) that could change wider-ranging spending habits would be able to mitigate some of the rebound effects experienced.

Despite the fact that several comparable studies (Druckman et al. 2011; Chitnis, Sorrell, Druckman & Jackson 2012; Chitnis, Sorrell, Druckman & Firth 2012) use a savings component in their analysis, I decided against it for a number of reasons. First, the assumption of constant preferences would imply an instantaneous decline in their utility upon forgoing meat, which should be recouped within the same period by alternative consumption. Furthermore, the short-run framework limits the analysis to one single time period, in which – according to the weak separability assumption – the optimal intertemporal budget allocation has already been decided upon. From a more practical perspective, one could argue that the individuals will use the saved money in the future in the same marginal way as they would now, and thus the impacts of time-delayed expenditures should be taken into consideration in the same way as the immediate effects. Finally, introducing a savings component would ideally require an estimate of the energy and greenhouse gas intensity of capital investment as well, since the individuals' savings are – in functioning financial markets – expected to be used simultaneously for firms' and other individuals' investments, which can incur high environmental impacts depending on the project. Such values however have not yet been estimated in a Swedish context.

5. Data Sources and Preparation

In a first step, this chapter presents the data sources used for the analysis and the methodologies used to collect them. In a second step, it details how the data was prepared for model use; specifically, it outlines the choice rules employed for data cleaning, creating correspondence between environmental and expenditure data, and for the construction of the 'average' consumer.

5.1. Data sources

In order to correctly interpret the model results, it is imperative to understand the underlying datasets used. This section first focuses on the environmental impact intensity indicators constructed by Johansson et al. (2010). It then turns to the vegetarian consumption survey carried out by Haddad and Tanzman (2003), and the household expenditure survey provided by Statistics Sweden (SCB 2014).

5.1.1. Environmental impact indicator data

The present analysis focuses both on energy intensity and CO_2-equivalent emissions. It benefits from recent work done by the Swedish Defense Research Agency FOI (Johansson et al. 2010), which calculated the energy and greenhouse gas intensity of 192 goods and services typically consumed by Swedish households. Johansson et al. (2010) arrived at these intensities by adapting the computer-based modeling program 'Energy Analysis Programme' (EAP), developed at Groningen University in the Netherlands, to the Swedish production context.

The 'Energy Analysis Programme' is based on a life cycle analysis framework that calculates the energy and greenhouse gas emissions associated with the production of raw materials and packaging materials, the manufacturing, the transportation to the final point of sale, the distribution process, the use and the disposal of the final good. However, in the final intensity data, the direct electricity use of household goods was excluded in order to avoid double-counting, since the household expenditure data also includes expenditures on electrical energy that captured this energy use. Figure 3 shows a graphical representation of the life cycle analysis framework. An important part of the EAP consists of an extensive database that includes information on products' make-up in terms of raw materials, their price and the tax associated with them, means and distance of transportation, which industrial sector it was produced in, which commercial sector the product is sold in, and how its disposal is managed. Through combining this information with data on the constitution of different industrial sectors' use of energy and fossil fuels, the program is able to calculate the indirect energy and greenhouse gas intensity of all disaggregated products and services (Johansson et al. 2010). In practice, it uses a combination of process analysis (for energy and greenhouse gas embedded in raw materials, packaging materials, transportation, and disposal) and input-output analysis (for the environmental impacts of manufacturing and sale) methodologies to arrive at these results.

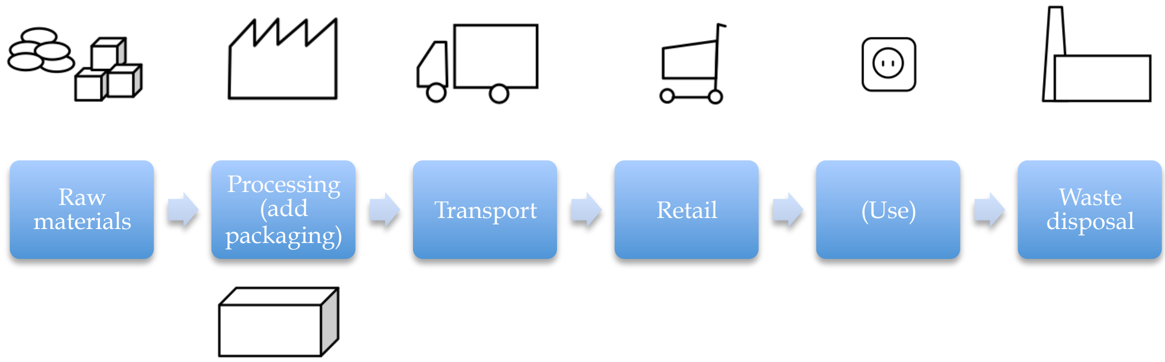

Figure 3: Life cycle analysis for environmental indicators. Source: Johansson et al. (2010)

The data was primarily based on production numbers from 2006, making it one of the most up-to-date databases accessible. It takes into consideration both direct energy use (required during the use of a product) and indirect energy use (required to manufacture, transport, sell and finally dispose of the product). This is indispensable for an accurate measurement of a household's energy footprint, since approximately 50% of households' energy needs are actually made up of indirect energy (Carlsson-Kanyama et al. 2005). Furthermore, the greenhouse gas intensity parameters take into consideration not only CO_2, but also methane (CH_4) and nitrous oxide (N_2O) emitted during the production process of all goods (translated into CO_2-equivalence according to their Global Warming Potential), which is of particular importance when considering the climate change impacts of meat consumption.

Most of these goods and services categories correspond directly to the categories of the Swedish Household Expenditure Survey, the data of which is collected yearly by the Swedish Statistical Agency. However, some correspondences had to be adapted, the process of which is specified in Section 5.2.4.

5.1.2. Vegetarian dietary choice data

In their comparison of vegetarian and non-vegetarian diets, Haddad and Tanzman (2003) use data from 13 313 participants in two waves (1994–1996 and 1998) of the Continuing Survey of Food Intake by Individuals (CSFII). This survey gathered information on US participants' food consumption habits through the means of two separate 24-hour dietary habits. Respondents that answered "yes" to the question "Do you consider yourself to be a vegetarian?" and that did not report any meat or fish consumption during the recall represented 0.9% of the nationally representative sample (n = 120). Their recall data was then contrasted with that of omnivore respondents that ate meat during the days studied (94.2%, n = 12 543).

The authors of the study report that "compared with nonvegetarians whose recalls listed meat, self-defined vegetarians who reported no meat tended to consume more grains such as cereals, pasta, and rice; more legumes; and more vegetables, especially dark green and deep yellow vegetables, but not more of commonly eaten vegetables such as tomatoes, lettuce, green beans, green peas, or corn. Vegetarians who avoid meat also ate more fruit, citrus fruit and juice, and dried fruit. However, consumption patterns of vegetarians who reported no meat were not significantly different from those of nonvegetarians in intakes of milk, milk drinks, yogurt, cheese, fats and oils, salad dressings, and sugars and sweets" (Haddad & Tanzman 2003, p. 631S). Based on their results, I decided to scale up all rice and cereal categories in the Swedish survey, all fruit (except for berries), cabbage and root vegetables, "other" vegetables (which included legumes), and fruit juice in proportion to the values provided by Haddad and Tanzman (2003). In Section 7.6.1, on the other hand, we scale all items recorded in the American survey up and down to account for additional differences in vegetarian food preferences and test the sensitivity of our results to that choice.

5.1.3. Baseline household expenditure data

In order to remain consistent with the environmental data, this analysis used the Household Expenditure Survey of 2006, and in particular the data differentiated by income deciles available on the web-based database provided by Statistics Sweden (SCB 2014). The Expenditure Survey data itself was collected over the course of the year 2006 and covers 4 000 households with at least one person below the age of 79 that were randomly sampled from all those registered in the Record of the Total Population in Sweden (Berglind 2007). It represents annual household expenditure subdivided in 117 categories, 71 of which were food consumption categories.

The data was procured through a combination of telephone interviews (which collected information on infrequent purchases such as furniture, appliances and travel) and households' bookkeeping records (diaries) of all reoccurring expenses over the course of 14 consecutive days. Alternatively, households had the option of collecting receipts and sending these to Statistics Sweden. The households were randomly assigned to start their expenditure diaries on one of the 52 weeks of the year in order to account for the seasonal variation in total size and composition of household expenditure. Furthermore, some data was derived from Statistics Sweden's records. The different subcategories of consumed goods and services were constructed according to the international COICOP (Classification of Individual Consumption On Purchase) classification scheme in order to simplify comparisons at an international level.

Measuring errors could arise from the fact that the sample two week-expenditure data was multiplied by 26 to arrive at yearly expenditures, which might not accurately depict the annual spending behavior if the two recorded weeks were significantly different from the remainder of the year. Furthermore, the non-response rate was 50%, which is not unusual for a survey requiring this amount of time investment, but which could lead to measurement errors. To minimize such errors, Statistics Sweden has used an estimation technique based on calibrated weights that takes auxiliary information from register information into account (Berglind 2007).

5.2. Data preparation

In order to achieve accurate results in both the Engel curve estimation procedure and the later calculation of the rebound effect, we need to assure that all datasets are consistent. The following section details the steps undertaken with that goal in mind.

5.2.1. Creation of consistency in household expenditures

Statistics Sweden calculated the average household expenditures in each category by dividing the sum of the per-category expenditure over all households in the decile by the number of households in that decile. This means that the sum of the most disaggregated categories did not always correspond to the number given in the overarching category, and the sum of the overarching categories in turn did not correspond to the total average household expenditure. In addition, some of the subcategory information was missing. In order to arrive at a consistent estimator of the marginal expenditure shares and the rebound effect, I thus adapted the data as follows: In a first step, I identified missing values and used the overarching category information to fill in the missing cells. If there was more than one value missing, I divided the excess money in the overarching category equally between the missing values. In a second step, I corrected all overarching category values to reflect the sum of the disaggregated categories, and the total expenditure to reflect the sum of the overarching categories. This allowed me to have a consistent tabulation of each decile's average household expenditure. This procedure is exemplified in Table 1.

Table 1: Example of data adjustment process

Category	First decile: before	First decile: after	Third decile: before	Third decile: after
1.6. Fruit and berries	1400	1400	1960	1970
1.6.1. Apples	140	140	270	270
1.6.2. Pears	130	130	60	60
1.6.3. Bananas	240	240	420	420
1.6.4. Citrus fruit	200	200	320	320
1.6.5. Other fresh fruit	260	260	210	210
1.6.6. Dried fruit and berries, nuts	200	200	280	280
1.6.7. Berries	150	150	330	330
1.6.8. Fruit and berry conserves	n/a	40	40	40
1.6.9. Fruit and berries - other	n/a	40	40	40

5.2.2. Creation of per capita expenditure values

Within the survey, the unit of analysis was determined to be a "housekeeping unit", defined as a "group of people who live together and have a common economy so that the various persons expenditure can not, in a meaningful way, be separated" (Berglind 2007, p. 8). Each of these housekeeping units was attributed a number of consumption units according to its make-up in order to account for the fact that expenditures do not increase proportionally with the number of individuals present in a household. Statistics Sweden's consumption unit scale 2004 attributes the weight of 1 to single households or the first adult of the household; 0.51 to the second adult; 0.6 to any additional adult after that; 0.52 to the first child (between the age of 0 and 19); and 0.42 to any additional child (Berglind 2007). Dividing the average expenditures per housekeeping unit in a particular decile by the average number of consumption units thus allows us to calculate the average expenditures per full consumption unit, which is then comparable across different household sizes. This is of particular importance since the average household size increased with increasing average expenditures, as shown below in Table 2. Using household data to derive marginal expenditure shares could thus have suffered from the spurious influence of different consumption patterns between large and small families; and controlling for the number of consumption units in the regression framework would introduce severe multicollinearity. Therefore, calculating expenditures per consumption unit was seen to be the theoretically soundest procedure for the purposes of comparability and reliability; and the remainder of the study will be based on per person – as operationalized by the consumption unit – data.

Table 2: Conversion of households into consumption units

Decile	Average expenditure/household [SEK/year]	Average number of consumption units/ household [SEK/year]	Average expenditure/ consumption unit [SEK/year]
1	125 950	1.06	118 820.75
2	139 120	1.10	130 018.70
3	183 940	1.17	157 213.68
4	189 730	1.29	147 077.52
5	227 490	1.42	160 204.23
6	262 350	1.65	159 000.00
7	290 170	1.86	156 005.38
8	314 410	1.95	161 235.90
9	382 240	2.09	182 889.95
10	439 240	2.12	207 188.68

5.2.3. Creation of average representative consumer data

Once I adapted the data as described, I also had to recalculate the average expenditure of my new unit of analysis – the individual consumption unit. I did this by multiplying the number of household observations in each decile with the corresponding average amount of consumption units in order to get an estimate of the 'total' consumption units both differentiated by decile and summed. I then weighed each category's decile-level observations by the amount of consumption units in that decile in order to get the mean over all consumption units. This can best be explained in the formula:

$$x_i^{Original} = \frac{\sum_{j=1}^{10}(x_{ij} * pop_j)}{\sum_{j=1}^{10} pop_j} \tag{13}$$

in which the subscript i denotes the subsector (with n subsectors in total);
the subscript j denotes the income decile;

x_{ij} denotes the decile-specific subsector expenditure derived from the data retrieved from Statistics Sweden [in SEK/person/year];

and pop_j denotes the observations in each decile, multiplied by the average number of consumption units, which should allow us to estimate the number of adult-equivalent consumption units and therefore persons in each decile.

We can then also calculate the total expenditure in the baseline scenario as

$$x_{tot}^{Original} = \sum_{i=1}^{n} x_i^{Original} . \tag{14}$$

The results of these calculations thus constitute the baseline expenditure scenario that will in my analysis reflect the status quo consumption pattern of an average Swedish consumer. In order to arrive at this representative individual's environmental footprint, we need to connect this information to the energy and greenhouse gas intensity parameters discussed above.

5.2.4. Correspondence between expenditure and environmental data

The intensities are expressed in megajoules (MJ) and kilograms of CO_2-equivalent emissions (kg CO_2-eq) per Swedish crown spent in the respective categories, making the calculation of total household energy use as straightforward as multiplying the category-related expenses by the respective parameters and summing over all categories. Mathematically,

$$E = \sum_{i=1}^{n} x_i * \gamma_i \tag{15}$$

in which the subscript i denotes the subsector (with n subsectors in total);

x_i denotes the scenario-specific expenditure in each subsector [in SEK/person/year];

and γ_i denotes the sector-specific intensities [in MJ/SEK or kg CO_2-equivalent/SEK respectively].

In the Household Expenditure Survey data, n = 117, with 71 food categories and disaggregated categories in 12 other broad sectors ranging from clothes to housing and entertainment. Despite the fact that the Swedish Defense Research Agency modeled their 192 intensity parameters to correspond broadly to the consumption categories surveyed by Statistics Sweden, there was not a 100% overlap. In the cases where there was no obvious match, I adopted the intensity parameters from the category closest to the one in question, and occasionally averaged over further disaggregated subcategories. Table 3 gives an example of my choice rule. Appendix 10.1 provides a full overview.

Table 3: Correspondence between categories

Category	Energy	GHG	Adaptations
1.4. Milk, cheese, and eggs			
1.4.1. Whole milk (Fat >=1,5%)	1.18	0.22	
1.4.2. Low-fat and skim milk (Fat <1,5%)	1.18	0.22	used ‘milk > 1,5 percent’
1.4.3. Powdered and canned milk	0.93	0.20	used ‘other milk products’
1.4.4. Thick milk and yoghurt	0.96	0.14	used average of sweetened and unsweetened yoghurt
1.4.5. Cream, sour cream, crème fraiche (Fat >=29%)	0.93	0.20	used ‘other milk products’
1.4.6. Cream, sour cream, crème fraiche (Fat <29%)	0.93	0.20	used other milk products’
1.4.7. Cheese (Fat >=17%)	1.10	0.23	used ‘hard cheese’
1.4.8. Cheese (Fat <17%)	1.10	0.23	used ‘hard cheese’
1.4.9. Eggs	0.82	0.08	
1.4.10. Milk, cheese and eggs - other	0.93	0.20	used ‘other milk products’

5.2.5. Creation of baseline scenario

Following this procedure, and adapting equation (15) using the baseline average expenditure values, I thus arrived at $E^{Original}$, the original energy use and CO_2-equivalent emissions incurred by the consumption behavior of an average Swedish consumer:

$$E^{Original} = \sum_{i=1}^{n} x_i^{Original} * \gamma_i \, . \tag{16}$$

Additionally, it was possible to calculate and compare the environmental impacts of individual consumption in all 10 deciles. This presents us with all data needed to implement the model as specified in the next chapter.

6. Model

Section 6.1 first presents summary statistics and results regarding the baseline scenarios and environmental indicators. Thereafter, the following sections specify the modeling of the vegetarian diet scenario, discuss the estimation and selection of the marginal expenditure shares, and describe the steps followed to arrive at the rebound effect.

6.1. Summary statistics

The datasets identified, as well as their combination, already allow for a rich analysis of the current consumption patterns and their environmental load. This section discusses the environmental intensity of different consumption goods in comparison, presents the environmental footprint of an average Swedish consumer according to this data, and compares both expenditures and footprints across income deciles.

6.1.1. Comparison of environmental intensity between categories

A first investigation of the life cycle analysis data furnished by the Swedish Defense Research Agency (Johansson et al. 2010) confirms the relevance of studying dietary changes and their contributions to sustainable lifestyles. Taking the means of all intensity parameters in the corresponding subcategories yielded the average energy and GHG intensity data in Table 4. It is apparent that food products have some of the highest energy needs per Swedish crown spent. Only consumables, living expenses and transportation expenditures are connected with higher per crown energy use. Food even tops all other categories in its greenhouse gas intensity, with only the transportation category coming close. The lowest energy intensities per expenditure are found in the consumption of tobacco, services, alcoholic beverages, and out-of-house dining. These are interesting results, but need to be carefully interpreted, particularly in relation to the regional context: Alcohol and tobacco are some of the most highly taxed goods in Sweden (Skatteverket 2012), and are relatively expensive on a per-weight basis. Thus, these expenditure-related results do not necessarily indicate that the production of tobacco is the most environmentally benign of all production processes, but could also be an indication that this is a category where Swedish consumers spend the most money in relation to the good received in return. Independent of the interpretation, we can conclude that rebound effects are expected to be limited if money saved is re-spent on categories such as services, leisure and culture activities, and highly taxed luxury goods such as alcohol and tobacco, whereas we expect a larger rebound effect if re-spending occurs mainly for housing and transportation purposes.

Table 4: Mean environmental intensity by category

Category	Energy intensity [MJ/SEK]	GHG intensity [kg CO_2-eq/SEK]
1. Food	0.815	0.082
2. Non-alcoholic beverages	0.607	0.041
3. Restaurant visits	0.490	0.011
4. Alcoholic beverages	0.443	0.029
5. Tobacco	0.100	0.004
6. Consumables	0.983	0.030
7. Services	0.367	0.008
8. Clothes and shoes	0.721	0.027
9. Housing (rent, energy)	1.763	0.044
10. Furniture	0.690	0.023
11. Healthcare	0.760	0.018
12. Transportation	1.270	0.078
13. Leisure and culture	0.574	0.027

When looking into food subcategories, the results from the literature review in Section 2.4 are supported by the Swedish data. Meat, dairy and fish products have energy and greenhouse gas intensities above the food average, while fruit and vegetables have some of the lowest intensity parameters. It is however also worthy to note that more processed goods generally have a higher environmental load, whether they were derived from animal- or plant-based primary ingredients, and that thus vegetarian options such as cereals or sweets could indeed be substitution options that increase, rather than decrease, the environmental footprint of the individual's diet.

Table 5: Mean environmental intensity by food category

Category	Energy intensity [MJ/SEK]	GHG intensity [kg CO_2-eq/SEK]
1. Food	0.815	0.082
1.1. Bread, cereal products	0.881	0.047
1.2. Meat	0.854	0.090
1.3. Fish and seafood	1.065	0.095
1.4. Milk, cheese and eggs	1.006	0.192
1.5. Oils and fats	1.188	0.092
1.6. Fruit and berries	0.404	0.041
1.7. Vegetables	0.462	0.047
1.8. Sweets, sugar	1.005	0.066
1.9. Sauces, dressings, condiments	0.910	0.058
1.10. Salt and spices	0.790	0.026
1.11. Baking powder, bullion, etc.	0.790	0.026
1.12. Snacks	0.780	0.036
1.13. Other foods	0.816	0.083

6.1.2. Expenditure pattern of baseline scenario

When estimating the expenditure pattern of an average Swedish individual consumer (operationalized by one consumption unit) as explained in Section 5.2.3, I arrived at an average expenditure of 163 828 Swedish crowns per year. As apparent in Figure 4, the bulk of this expenditure occurs in the housing, transportation, and leisure categories. Food expenditures constitute 12% of total spending. The 'other' category includes all categories with less than 2% of total spending: beverages (alcoholic and non-alcoholic), restaurant visits, tobacco, consumables, and healthcare.

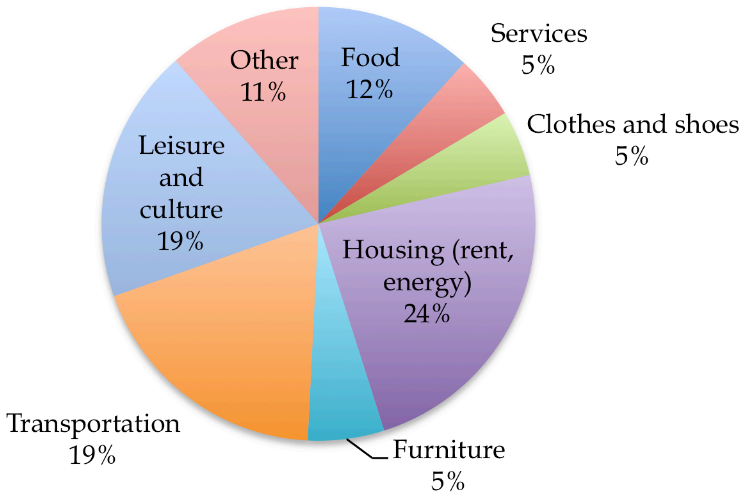

Figure 4: Expenditure pattern of average consumer. Source: SCB (2014), author's calculation.

6.1.3. Environmental footprint of baseline scenario

After multiplying each category's average expenditure with the environmental intensity data, we can construct baseline environmental footprints both concerning energy use and GHG emissions linked to the current average lifestyle. According to this data, the typical Swedish consumer uses 196 669 MJ of energy and emits 9 383 kg CO_2-equivalents per year. These results are close to those of Reinders et al. (2003), whose estimates for the Swedish household energy footprint was 328GJ/household/year in 1994. At an average of 1.59 consumption units per household (albeit in 2006), this corresponds to 206 GJ/capita/year. The calculations are also well in line with approximations of the Swedish Environmental Agency, which estimates that private consumption creates between 9.6 tons and 10 tons of CO_2 per capita annually (Hjerpe et al. 2013).

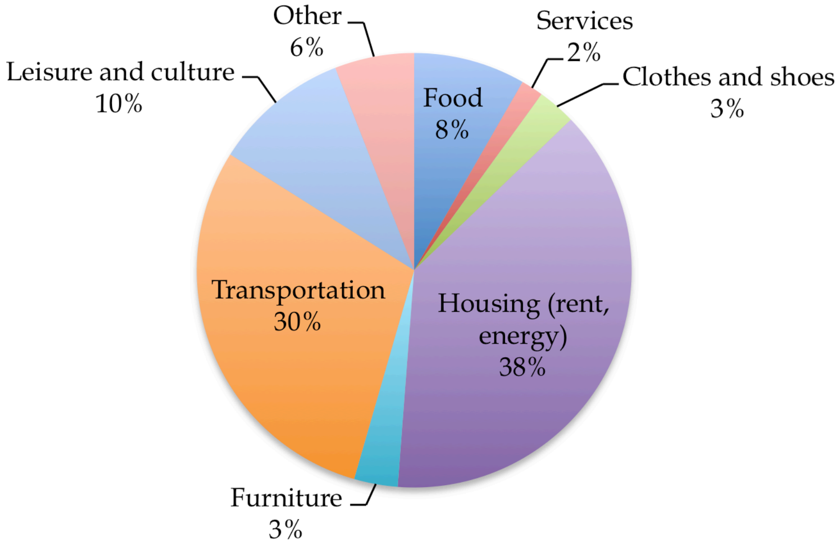

Figure 5: Energy footprint of average consumer. Source: SCB (2014), Johansson et al. (2010), author's calculation.

As Figure 5 shows, most energy use comes from Swedish consumers' housing and transportation consumption. The food category has an even smaller percentage-wise impact than the expenditure data would suggest, at only 8% of total energy use. This result mirrors findings of Reinders et al. (2003) of the average EU footprint, where a food expenditure share of 20% only accounted for 18% of energy use, and qualifies the suggestion suggested in previous literature (Goedkoop et al. 2002; Berners-Lee et al. 2012) that food consumption is overproportionally environmentally harmful. In terms of effectiveness, improving consumption habits regarding car use or heating, for instance, might in the Swedish context lead to larger overall savings.

On the other hand, the results by Berners-Lee et al. (2012) are mirrored by the greenhouse gas footprint of the average Swedish consumer in Figure 6. Here, food consumption does indeed have an overproportionally high impact on the environment, contributing almost as many greenhouse gases as the housing category, though it is not the sector with the largest proportional effect – that is still the transportation sector.

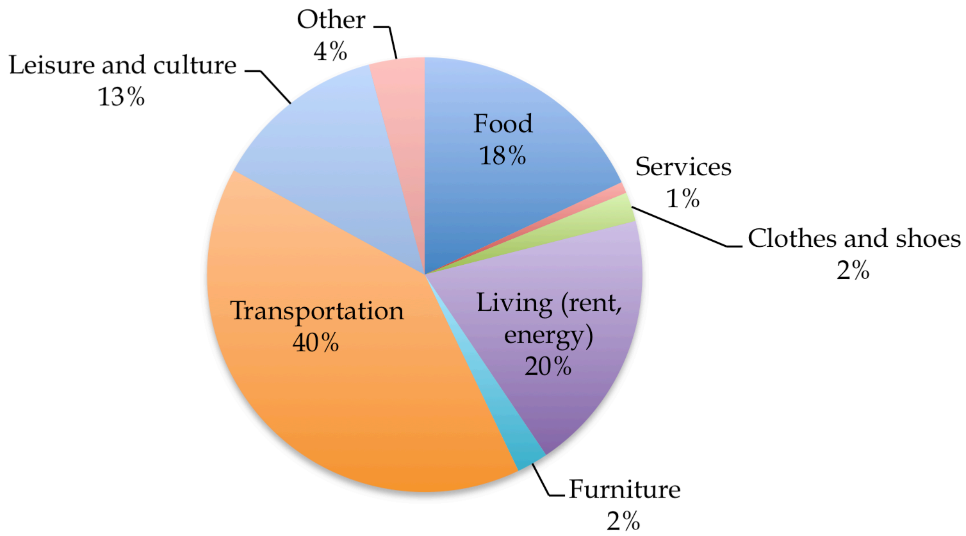

Figure 6: GHG footprint of average consumer. Source: SCB (2014), Johansson et al. (2010), author's calculation.

6.1.4. Comparison of expenditure patterns between incomes

When comparing expenditure patterns across deciles, several points are noteworthy. First, after equalizing expenditures according to the number of consumption units in each household, there is a need to rearrange the ordering from lowest to highest per unit expenditure. This finding justifies the choice of creating a per-capita baseline scenario, since the analysis would otherwise conflate household size and per-capita wealth. The graphical analysis of absolute expenditure after this adaption process also shows that per-capita wealth especially of the middle-income segments of Swedish society is extremely equal, with barely notable per-capita increases in expenditure between the third all the way to the eighth household decile.

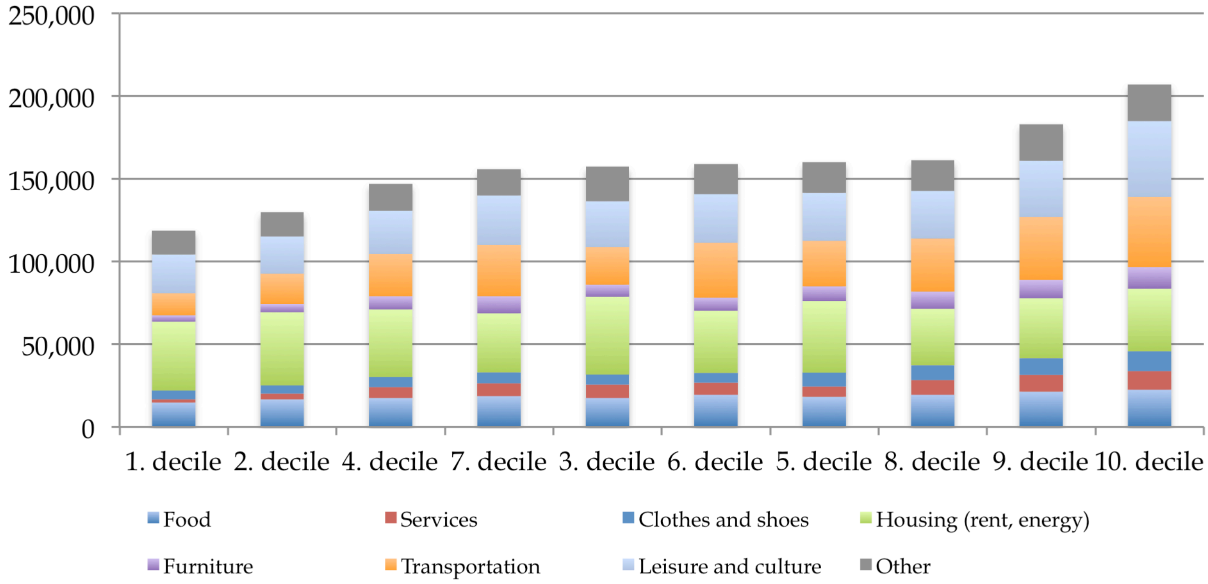

Figure 7: Absolute per category expenditure by income, in SEK/capita/year. Source: SCB (2014), author's calculation.

Furthermore, when comparing proportional spending patterns we can note several trends. First, Engel's law (Zimmerman 1932) – that proportional expenditure on food decreases with increasing income – is only weakly present in the Swedish case, as the highest income categories spend almost equal proportions of their income on food (12 – 13%) as the lowest ones (14 – 15%). Then, we can identify a shift toward services, transportation, and leisure expenditure, and away from housing, as incomes increase. This is in line with theory, as essential needs such as housing are addressed first and luxury wants are prioritized later (Zimmerman 1932).

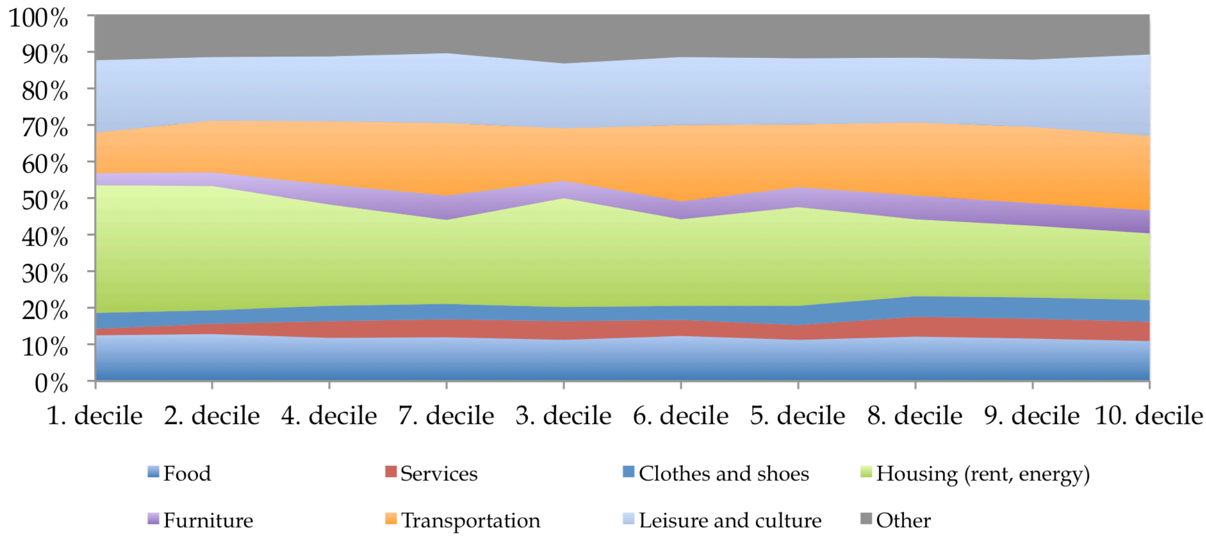

Figure 8: Proportional per-capita expenditure. Source: SCB (2014), author's calculation.

6.1.5. Comparison of environmental footprints between incomes

Finally, a comparison of the environmental footprint data between the income categories shows that while impacts clearly increase with increased income, the relationship is more complex: ordered from lowest to highest expenditure, footprints in the energy or greenhouse gas categories do not linearly increase, but vary across deciles. This confirms that the type of consumption behavior, as well as the total magnitude of consumption, will influence the environmental sustainability of the particular consumer.

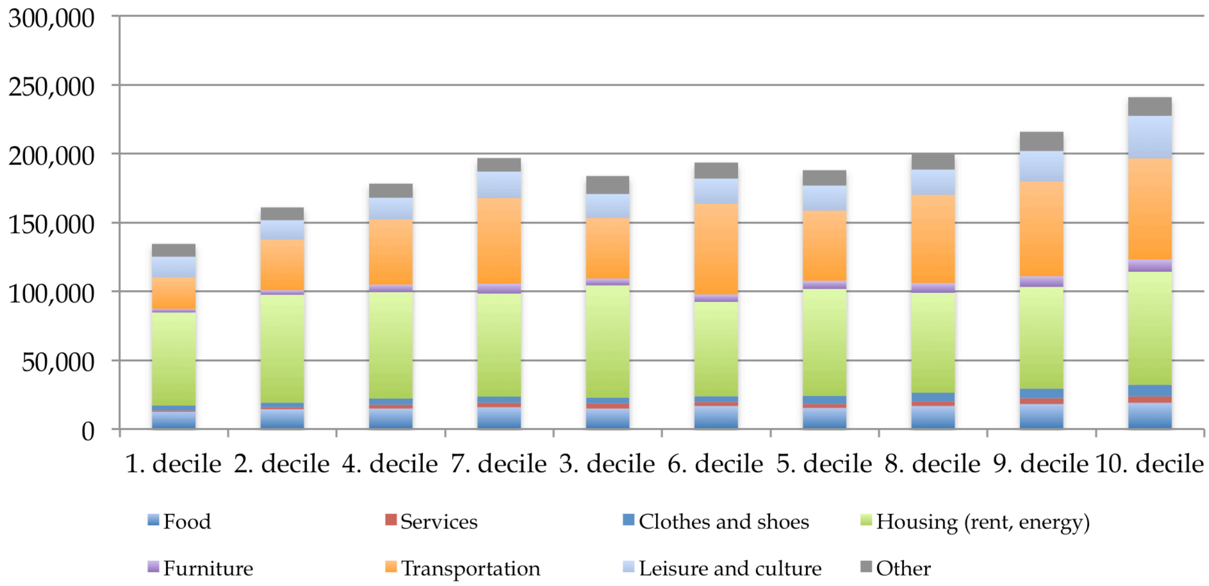

Figure 9: Energy use by income, in MJ/capita/year. Source: SCB (2014), Johansson et al. (2010), author's calculation.

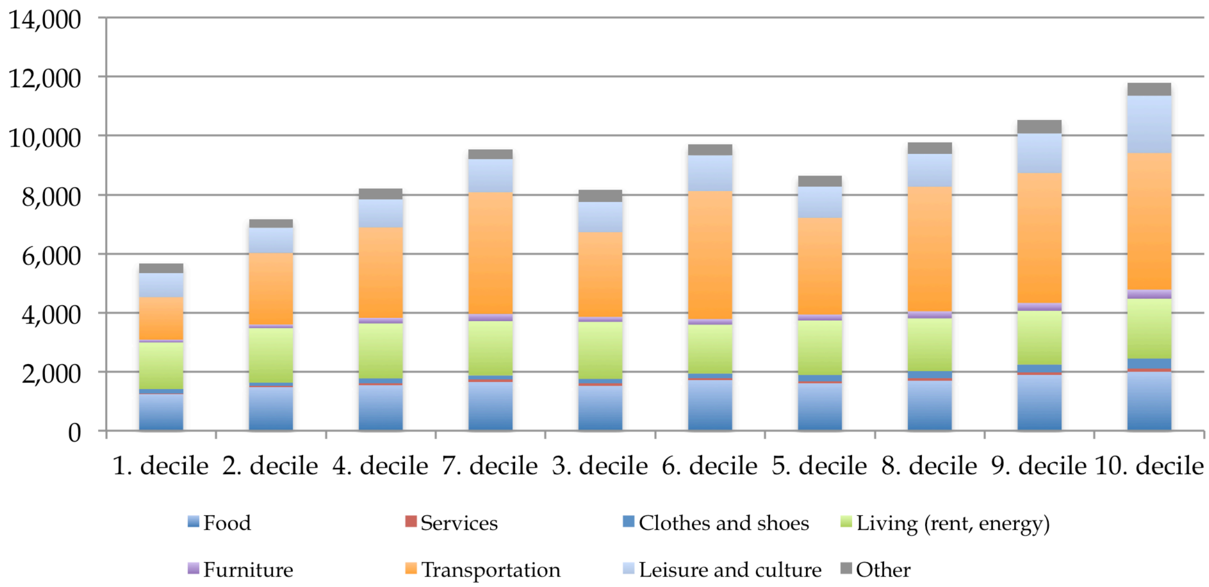

Figure 10: Greenhouse gas emissions by income, in kg CO2-eq/capita/year. Source: SCB (2014), Johansson et al. (2010), author's calculation.

6.2. Model specifications

This section specifies the steps followed in order to arrive at the results discussed in Chapter 7. It first describes the estimation of the marginal expenditure shares used and then describes the procedure used to arrive at the first and second round environmental footprints. Finally, it restates the method used to calculate the rebound effects.

6.2.1. Derivation of marginal expenditure shares

As explained in Sections 4.3.3 and 4.4, I tested linear, quadratic, Working-Leser and double semi-log specifications of the Engel curves for all 117 subcategories provided in the Household Budget Survey, and tested both OLS and WLS estimation methods.

Thus, in practice, I regressed the subsector expenditure on the total expenditure, and if applicable the total expenditure squared or the natural logarithm of total expenditure, according to the following equations:

Linear Model

$$x_{ij}^{Original} = \alpha_i + \beta_i * \left(\sum_{i=1}^{n} x_{ij}^{Original}\right) + \varepsilon_{ij} \,. \qquad (17)$$

Quadratic Model

$$x_{ij}^{Original} = \gamma_i + \vartheta_i^{Linear} * \sum_{i=1}^{n} x_{ij}^{Original} + \vartheta_i^{Quadratic} * \left(\sum_{i=1}^{n} x_{ij}^{Original}\right)^2 + \varepsilon_{ij} \,. \qquad (18)$$

Working-Leser Model

$$\frac{x_{ij}^{Original}}{\sum_{i=1}^{n} x_{ij}^{Original}} = \mu_i^{Linear} + \mu_i^{Log} * ln\left(\sum_{i=1}^{n} x_{ij}^{Original}\right) + \varepsilon_{ij} \,. \qquad (19)$$

Double Semi-Log Model

$$x_{ij}^{Original} = \delta_i + \theta_i^{Linear} * \sum_{i=1}^{n} x_{ij}^{Original} + \theta_i^{Log} * \ln\left(\sum_{i=1}^{n} x_{ij}^{Original}\right) + \varepsilon_{ij} \,. \qquad (20)$$

in which the subscript i denotes the subsector (with n subsectors in total);
the subscript j denotes the income decile;
x_{ij} denotes the decile-specific subsector expenditure per consumption unit derived from the data retrieved from Statistics Sweden [in SEK/person/year];
$\sum_{i=1}^{n} x_{ij}$ denotes the decile-specific total expenditure per consumption unit [in SEK/person/year];
and β_i, ϑ_i^{Linear}, $\vartheta_i^{Quadratic}$, μ_i^{Linear}, μ_i^{Log}, θ_i^{Linear} as well as θ_i^{Log} define the relationship of the sector-specific expenditure to total income.

I used the population size in consumption units (which I calculated by multiplying the number of households in each decile by the respective average amount of consumption units) as weights for the Weighted Least Squares method in order to improve focusing accuracy for the estimations, following Haque (2005).

We can therefore calculate the marginal expenditure share (or marginal propensity to spend) as the slope of sector-specific expenditure graphed against total income in four different ways:

(i) as the simple coefficient $\beta_i^{Original}$ from the linear regression;

(ii) using the results of the quadratic regression as follows:

$$\beta_i^{Original} = \vartheta_i^{Linear} + 2 * \vartheta_i^{Quadratic} * \sum_{i=1}^{n} x_{ij}^{Original} ; \tag{21}$$

(iii) using the results from the Working-Leser regression as follows:

$$\beta_i^{Original} = \mu_i^{Linear} + \mu_i^{Linear} * \left[1 + ln\left(\sum_{i=1}^{n} x_{ij}^{Original}\right)\right]; \tag{22}$$

(iv) using the results from the double semi-log regression as follows:

$$\beta_i^{Original} = \theta_i^{Linear} + \frac{\theta_i^{Log}}{\sum_{i=1}^{n} x_{ij}^{Original}} . \tag{23}$$

It is thus evident that methods (ii), (iii) and (iv) will yield different marginal expenditure shares for different income deciles, while method (i) yields equal shares across all incomes. Table 6 shows an example of the results of the different approaches – using weighted least squares with population size as weighting factor – in comparison for the average consumer. The WLS results were very close to the OLS regression values, as can be seen in the comparison of the Working-Leser and double semi-log results as well as in a full overview of results in Appendix 10.2. It is noteworthy that there exist few differences between the linear, quadratic, Working-Leser and double semi-log results; and that the marginal values are significantly different from the average consumption shares, supporting the case that using average shares for a redistribution of expenditure may lead to flawed conclusions.

Table 6: Example of marginal expenditure share results

Category	Average shares	WLS Linear	WLS Quadratic	OLS Working-Leser	WLS Working-Leser	OLS Double Semi-Log	WLS Double Semi-Log
9. Housing							
9.1. Rent/charges for housing	0.116	-0.307	-0.325	-0.350	-0.330	-0.315	-0.311
9.2. Repairs	0.019	0.042	0.047	0.051	0.047	0.043	0.043
9.3. Home and house insurance	0.007	0.006	0.006	0.005	0.006	0.006	0.006
9.4. Housing services	0.015	0.021	0.023	0.025	0.023	0.022	0.022
9.5. Electricity, gas and fuels	0.036	0.038	0.040	0.043	0.040	0.039	0.039
9.6. Mortgages	0.045	0.134	0.134	0.132	0.134	0.134	0.133

The adding-up restriction, also known as the Engel aggregation condition, requires that all expenditure is accounted for and that there is no excess expenditure above and beyond the increase in income (Deaton & Muellbauer 1980; Chitnis, Sorrell, Druckman & Jackson 2012). This implies the following condition:

$$\sum_{i=1}^{n} \beta_i^{Original} \equiv 1 \,.$$

When summing the estimated values, we get the following results:

Table 7: Engel aggregation condition in practice

	Average shares	WLS Linear	WLS Quadratic	OLS Working-Leser	WLS Working-Leser	OLS Double Semi-Log	WLS Double Semi-Log
Average consumer	1	1.000160	1.000127	1.000098	1.000119	1.000385	1.000154
1. decile	1	1.000160	0.999777	1.000070	1.000081	1.001861	0.999713
10. decile	1	1.000160	1.000465	1.000119	1.000146	0.999570	1.000398

In practice, this holds rather well for the average households as well as for decile-specific estimations as Table 7 shows. The slight deviations from the aggregation condition are within the margin of error and in practice only imply re-spending of one Swedish crown more or less compared to the baseline scenario.

Due to the inherent differences in total expenditures, I used heteroskedasticity-corrected error terms. Since I only have 10 semi-aggregated observations and no access to micro-data, the estimated coefficients were not always significant. However, since this data is used to model *ceteris paribus* behavior rather than for explanatory purposes, this seems acceptable.

As the regressions had to be estimated for 117 separate categories, standard goodness-of-fit tests were of limited use in model choice. A visual inspection of the predictive power of the models found that all models were comparable when observed data approached linearity such as in Figure 11. However, both the quadratic and the double semi-log models fit the observed data better in most instances because they allowed for stronger non-linearity than the linear and Working-Leser forms. This is visible for example in Figure 12 and Figure 13; the full graphical overview can be provided upon request. Comparing the two, the double semi-log form estimated by Weighted Least Squares found more theoretical support (refer to Section 4.3.3 and Haque (2005) for a more in-depth discussion). Thus, I decided to use the WLS double semi-log estimations for my baseline scenario; a comparison of results acquired using the other models will be provided under Section 7.6.2 in the sensitivity analysis.

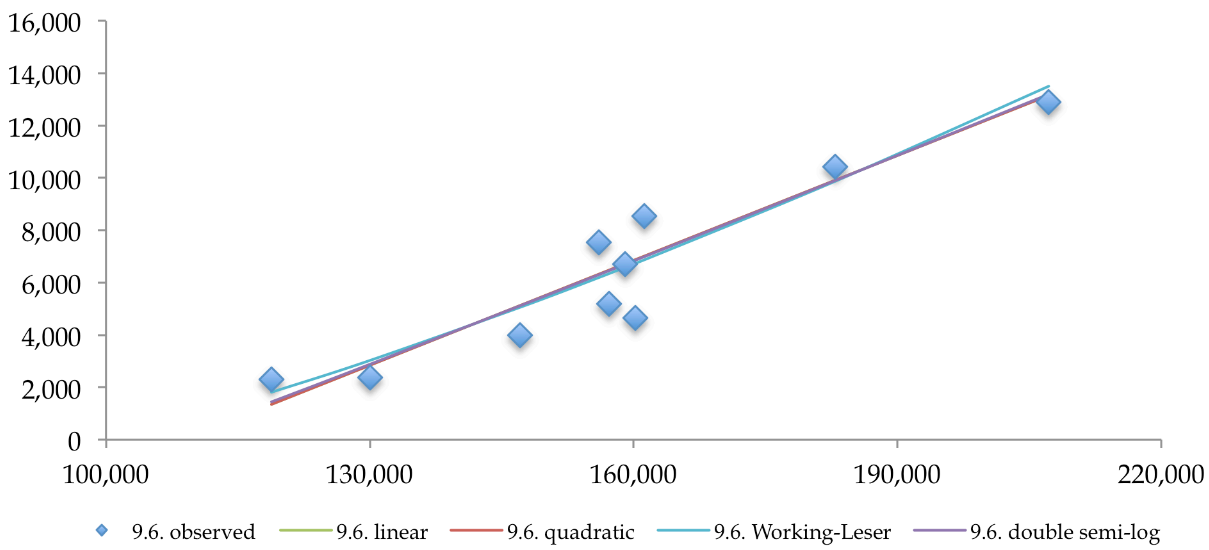

Figure 11: Estimations of Engel curves for Category 9.6. Mortgages. The X-Axis shows total per capita income in SEK/person/year, while the Y-Axis graphs the per category expenditure in SEK/person/year. Data source: SCB (2014).

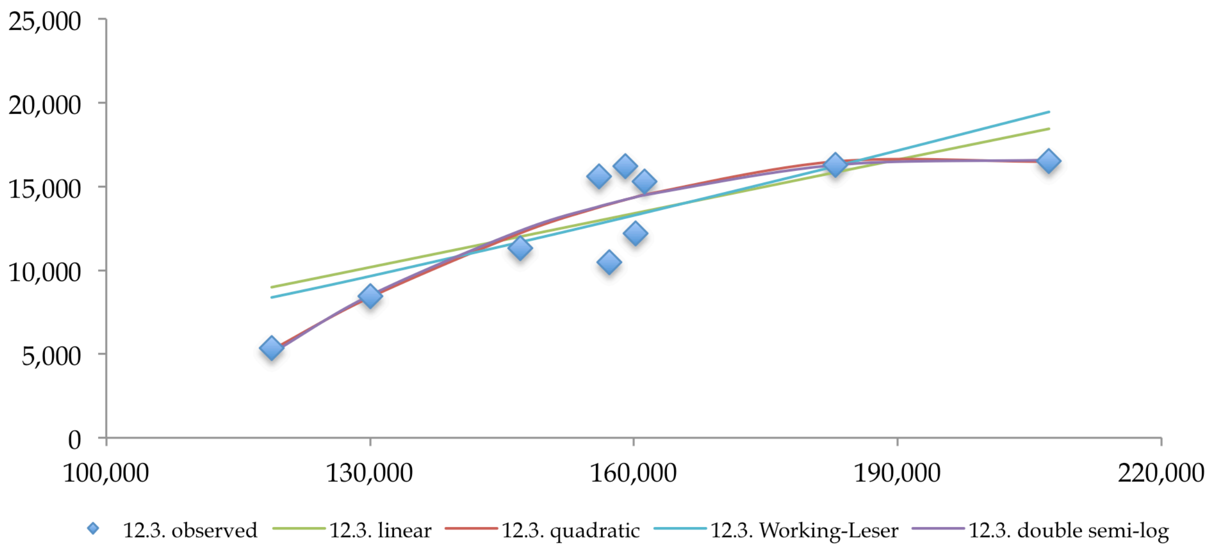

Figure 12: Estimations of Engel curves for Category 12.3. Car use. The X-Axis shows total per capita income in SEK/person/year, while the Y-Axis graphs the per-category expenditure in SEK/person/year. Data source: SCB (2014).

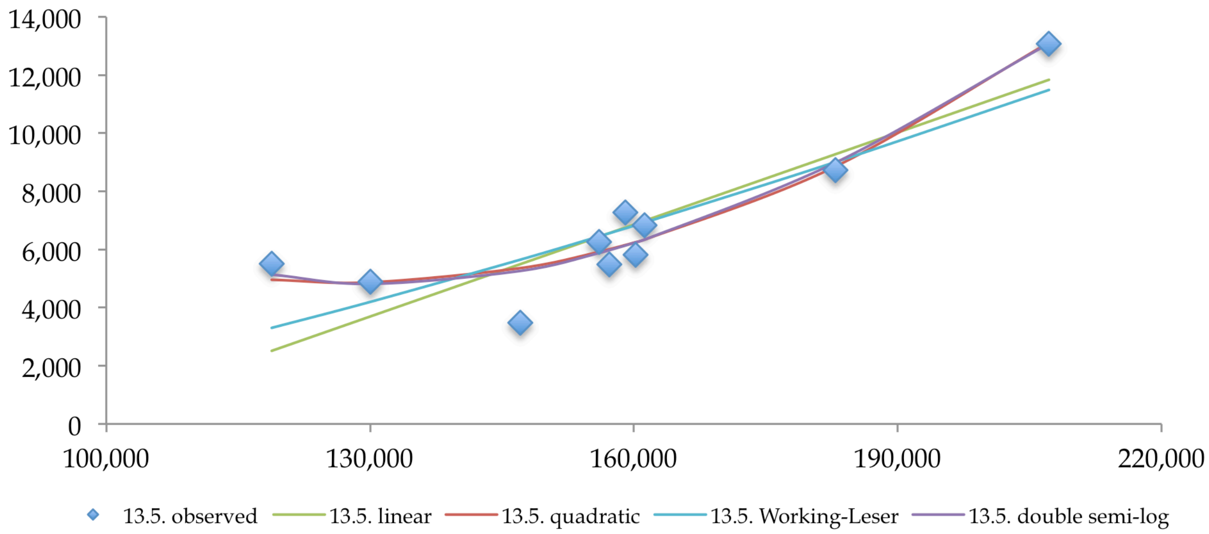

Figure 13: Estimations of Engel curves for Category 13.5. Travel, hotel. The X-Axis shows total per capita income in SEK/person/year, while the Y-Axis graphs the per-category expenditure in SEK/person/year. Data source: SCB (2014).

6.2.2. Vegetarian marginal expenditure shares

Having calculated the marginal expenditure shares $\beta_i^{Orriginal}$, we run into the problem that these marginals are also defined for the meat and fish consumption categories that we explicitly set to 0 to simulate a vegetarian consumption pattern.

In order to impose as few assumptions as possible, I followed Murray (2013) and redistributed the marginal values of meat or fish categories among all other consumption categories in proportion to the original distribution of the non-meat/fish marginal shares. In that case,

$$\beta_i^{Veg} = \begin{bmatrix} 0 & if\ i\ \in\ meat\ or\ fish\ category \\ \beta_i^{Original} + \frac{\beta_i^{Original}}{\sum_i^o \beta_i^{Original}} * \sum_i^p \beta_i^{Original} & otherwise \end{bmatrix} \tag{24}$$

in which the subscript i denotes the subsector;
the array o includes all non-fish/meat subsectors;
and the array p includes all fish/meat categories.

This transformation provided the marginal propensities to spend for a newly vegetarian consumer that I subsequently used to estimate the rebound effects. This next section will first detail the initial modeling scenario.

6.2.3. First round effects

The switch toward a vegetarian diet entails more than simply the elimination of meat and fish from the plate. An additional challenge is to guarantee an adequate nutritional profile of the 'new' vegetarian diets, in particular ensuring that individuals consume enough calories, protein and micronutrients once they stop eating meat and fish. To best model this change, I utilized the data from Haddad and Tanzman (2003), as described in Section 5.1.2, in order to create representative diets of vegetarians and non-vegetarians respectively. Since this data was given in mean intakes of g/person/year, and my data is in SEK/person/year, I used relative differences to scale up food consumption categories that were consumed in significantly higher quantities in the vegetarian sample diet. For instance, US vegetarians on average consumed 51g of rice per day, while non-vegetarians only consumed 23g. The vegetarians thus had a consumption that was 2.21 times higher; I therefore multiplied the expenditure in the 'rice' category by 2.21 for the first round effects. Mathematically, this step can be described as follows:

$$x_i^{First\ Round} = \begin{cases} 0 & if \quad i\ belongs\ to\ fish\ or\ meat\ category \\ \dfrac{cons_i^{Veg}}{cons_i^{Non-veg}} * x_i^{Original} & if \quad i\ is\ a\ substitution\ category \\ x_i^{Original} & otherwise \end{cases} \tag{25}$$

in which the subscript i denotes the subsector;

$cons_i^{Veg}$ denotes the sample consumption of the ith category in the representative US vegetarian diet [in g/individual/day];

$cons_i^{Non-veg}$ denotes the sample consumption of the ith category in the representative US non-vegetarian diet [in g/ individual/day];

and $x_i^{Original}$ denotes the average sector-specific expenditure of our baseline scenario [in SEK/household/year].

I chose to adapt those dietary items that Haddad and Tanzman (2003) reported as having been consumed to a significantly higher degree by vegetarians than by non-vegetarians and that could from a nutritional perspective be seen as substitutes to meet caloric or protein needs. Items included in the scaling effort were cereals (rice and pasta), select fruit and vegetables, nuts, and legumes such as lentils and beans.

This approach comes with several caveats. First, I am assuming that changes in dietary patterns in the United States can be used as a proxy for similar changes in Sweden. This might be a tenuous assumption if non-vegetarian or vegetarian diets are fundamentally different in the two countries. Then, as explained in the theoretical section, this approach was seen as the most likely to keep food preferences – except for meat products – constant within a short-term framework of analysis. However, this selectivity might be called into question and thus Section 7.6.1 provides a sensitivity analysis where all categories are adapted to test whether it makes a significant difference on the result.

The total expenditure related to this consumption pattern can thus be calculated as:

$$x_{tot}^{First\ Round} = \sum_{i=1}^{n} x_i^{First\ Round} \tag{26}$$

and the potential expenditure savings as:

$$savedx = x_{tot}^{Original} - x_{tot}^{First\ Round}\ . \tag{27}$$

Thereafter, we can apply the environmental intensity parameters as follows below:

$$E^{First\ Round} = \sum_{i=1}^{n} x_i^{First\ Round} * \gamma_i\ . \tag{28}$$

6.2.4. Second round effects

As motivated in Section 4.4, in the current scenario, I decided to assume full re-spending of the saved expenditure. According to our model, the average Swedish consumer would treat the saved expenditure just as additional income. Thus, the subsector-specific expenditures in the rebound sector can be defined as:

$$x_i^{Second\ Round} = x_i^{First\ Round} + \beta_i^{Veg} * savedx \tag{29}$$

which can be further disaggregated as:

$x_i^{Second\ Round}$ = 0 if $i \in$ fish or meat category;
= $x_i^{First\ Round} + \beta_i^{Veg} * [\sum_{i=1}^{n} x_i^{Original} - \sum_{i=1}^{n} x_i^{First\ Round}]$ otherwise.

This leads us to re-construct the original expenditure of 163 828 SEK. We can then apply the environmental intensity parameters as follows:

$$E^{Second\ Round} = \sum_{\mathrm{i}=1}^{n} x_i^{Second\ Round} * \gamma_i\ . \tag{30}$$

This provides us with all necessary values to arrive at our final rebound estimate, which can be calculated using the formula:

$$R = \frac{\Delta E_P - \Delta E_A}{\Delta E_P}\ . \tag{1}$$

7. Results and Discussion

This chapter reports the results of implementing the model specified in Chapter 6 and puts them into context, including comparisons with previous literature. Sections 7.1 to 7.4 focus on the main rebound scenario of an average consumer. Thereafter, Section 7.5 presents differences in results according to household wealth. Finally, 7.6 tests these results for their sensitivity to several alternative specifications and presents different re-spending scenarios that could be considered for policy recommendations.

7.1. First round environmental effects

Having constructed the new vegetarian diet and kept all other consumption constant during the first round of the behavior change, the new energy use footprint is 193 097 MJ/person/year, signifying a potential decrease of energy use of 3 572 MJ or 1.81% compared to the original consumption pattern. The GHG savings are slightly more significant, with a new footprint at 8 994 kg CO_2-eq/person/year, and potential savings of 4.15%, 390 kg CO_2-eq/person/year. Table 8 on page 61 shows that when only food and drink expenditure is considered, the savings are 16% and 20%. This is in line with hypothesis (H1) – there are indeed initial benefits of switching to vegetarianism. While these potential savings are small, they are not insignificant: summarized over the entire Swedish population, they amount to 33 994 724 GJ/year, or the equivalent of 2 551 km driven per person. In greenhouse gas terms, each Swede would even save the equivalent of 3 979 km driven per year if going vegetarian (Mäkelä 2012).

Yet, the results are smaller than those of comparable studies. Alfredsson's (2004) 'green diet' reduced energy requirements by around 5% and CO_2 emissions by about 13%. Discrepancies may arise due to the different data sources – Alfredsson's consumption data set stemmed from 1996 and she used Dutch environmental data. Furthermore, the diets were not completely identical. Hjerpe et al. (2013), on the other hand, state that a reduction of meat consumption by only 50% in Sweden would lead to almost equivalent reductions of 300 – 400 kg of CO_2-equivalents per person and year. One possible reason could be that they did not consider the impact of dietary replacements. These first insights already showcase the sensitivity of such estimates to the choice of data and methods.

In terms of comparability, one can note that Nordic per-capita consumption patterns are very similar to pan-European patterns, though Nordic consumers eat slightly less vegetables and slightly more dairy products than the average European (Fritsche et al. 2009). Meat consumption patterns are in line with the pan-European average (Fritsche et al. 2009); thus, the following results might indicate the direction of results in other European countries as well, though the specific estimations rely on country-specific consumption and production data and should be interpreted in that context.

7.2. First round expenditure effects

The new vegetarian expenditure calculated according to the specifications given in 6.2.3 is 160 728 SEK, 3 100 SEK less than the baseline expenditure. Therefore, hypothesis (H2) is also supported by the data: switching to a vegetarian diet could indeed save an average Swedish consumer money, namely an estimated 1.89% of their current annual budget or around 16% of their expenditure on food. These results are very similar to Alfredsson (2004) who found that a 'green' diet was 15% cheaper than the original. Lenzen and Dey's (2002) new diet was even 30% cheaper; however, they also decreased the caloric intake by 40%, while I strived to keep it comparable. As mentioned, the shift toward all-organic produce makes Carlsson-Kanyama et al.'s (2005) study a stand-alone case in the literature where a 'sustainable diet' was 10% more expensive than the initial one; this particular case is also investigated in more detail in Section 7.6.3 of the sensitivity analysis.

7.3. Second round environmental effects

Using the WLS double semi-log marginal values, adjusted for a vegetarian consumer as explained in Section 6.2.2, we find that the average consumer re-spends most of her savings in the transportation and leisure categories, while furniture, clothes and services also receive minor shares of the re-expenditure. Table 10 on page 62 gives an overview of all marginal values. Applying the environmental impact indicators to this new consumption basket, we arrive at new (adjusted) vegetarian environmental footprints of 196 529 MJ/person/year in terms of energy use and GHG emissions of 9 186 kg CO_2-equivalent/person/year. As Table 8 on page 61 shows, this leads us to the conclusion that the average consumer would barely save any energy compared to the baseline scenario and only save 2.1% of the original greenhouse gas emissions. These insights can be expressed more formally by using the environmental rebound concept.

7.4. Vegetarian rebound effects

To recapitulate, for the example of energy use we have calculated

$E^{Original} = \sum_{i=1}^{n} x_i^{Original} * \gamma_i$ = 196 669 MJ/person/year;

$E^{First\ Round} = \sum_{i=1}^{n} x_i^{First\ Round} * \gamma_i =$ 193 097 MJ/person/year;

and

$E^{Second\ Round} = \sum_{i=1}^{n} x_i^{Second\ Round} * \gamma_i$ = 196 529 MJ/person/year.

Inserting these values into the definition of the rebound effect

$$Rebound = \frac{\Delta E_P - \Delta E_A}{\Delta E_P} \tag{1}$$

in which

$$\Delta E_P = E^{Original} - E^{First\ Round}$$

and

$$\Delta E_A = E^{Original} - E^{Second\ Round}$$

to calculate the share of potential energy savings that would not come into effect due to the average re-spending behavior yields the result of

$$Rebound_{Energy} = \frac{(196\,669 - 193\,097) - (196\,669 - 196\,529)}{(196\,669 - 193\,097)} = 96.07\%\,.$$

Thus, almost all potential energy savings are replaced by energy use stemming from alternative expenditures, making a vegetarian diet an ineffective way to reduce energy use by individuals. However, applying the same formula to the greenhouse gas emissions attached to the different consumption patterns, we get the following result:

$$Rebound_{GHG} = \frac{(9\,383 - 8\,994) - (9\,383 - 9\,186)}{(9\,383 - 8\,994)} = 49.39\%\,.$$

Despite total re-spending of the initial budget constraint, consumers can thus still make a difference in their CO_2 footprint by changing their consumption patterns. Still, hypothesis (H3) is strongly supported by these findings: rebound effects of vegetarian diets are clearly a factor that should be taken into consideration. These results are very much in line with previous research: greenhouse gas rebound effects of diet changes range from 50% (Lenzen & Dey 2002) to 59% (Druckman et al. 2011), while energy rebound effects are generally higher, spanning from 111 – 123% (Lenzen & Dey 2002) to 200% (Alfredsson 2004).

Table 8: Results of average consumer rebound scenario

	Energy use [MJ/person/year]	GHG emissions [kg CO_2 – eq/person/year]
Baseline scenario	196 669	9 383
First round footprint	193 097	8 994
First round % change	-1.81%	-4.15%
First round food % change	-16.64%	-20.46%
Second round footprint	196 529	9 186
Second round % change	-0.07%	-2.10%
Second round food % change	-15.35%	-19.41%
Rebound effect	96.07%	49.39%

Note: "food % change" reflects the percentage change in the environmental load of food and drink consumption.

7.5. Household wealth effects

We have established that individuals are likely to have different re-spending behaviors, and therefore different marginal expenditure shares, which depend on their household's income. To investigate whether these differences have a strong influence on the rebound results, we can first compare the marginal expenditure shares for select categories between the first, average, and last decile, and in a second step compare the resulting rebound effects.

Table 9: Comparison of marginal expenditure shares across income deciles

Category	First decile	Average consumer	Tenth decile	Energy intensity	GHG intensity
5. Tobacco	0.0043	-0.0077	-0.0142	0.10	0.004
9.6. Mortgages	0.1281	0.1335	0.1364	1.20	0.026
12.3. Car use	0.3469	0.1114	-0.0187	3.25	0.235
13.5. Travel, hotel	-0.0550	0.1030	0.1896	0.83	0.059

In Table 9, we can see that for some inferior items, such as tobacco, expenditure increases with income at lower wealth levels but will decrease for higher levels of income. Other items, such as mortgages, have relatively constant expenditure shares, whereas other marginal expenditure shares consistently decrease with increasing income. If these are items with higher than average energy or GHG intensity, such as car use, rebound effects are expected to be higher for lower income deciles. We expect a similar effect if items with relatively small environmental impacts increase in their marginal expenditure shares with wealth, such as category 13.5. This tendency holds true for most expenditure categories, as the semi-aggregated comparison of the adjusted vegetarian marginal expenditure shares in Table 10 shows. In fact, we find that the marginal expenditure on leisure activities, while taking the lion share of re-expenditure for the tenth decile, is even negative for the first decile, while the majority of their spending flows toward transportation, the category with the second-highest energy and highest greenhouse gas intensity. This leads us to expect important differences in size of the rebound effect across income deciles.

Table 10: Vegetarian marginal expenditure shares across income deciles

Category	First decile	Average consumer	Tenth decile	Energy intensity	GHG intensity
1. Food	0.082	0.061	0.049	0.770	0.078
2. Non-alcoholic beverages	0.001	0.003	0.004	0.607	0.041
3. Restaurant visits	0.071	0.070	0.070	0.490	0.011
4. Alcoholic beverages	0.033	0.009	-0.004	0.443	0.029
5. Tobacco	0.004	-0.008	-0.015	0.100	0.004
6. Consumables	0.031	0.020	0.014	0.983	0.030
7. Services	0.199	0.098	0.043	0.367	0.008
8. Clothes and shoes	0.053	0.097	0.121	0.721	0.027
9. Housing (rent, energy)	-0.210	-0.070	0.007	1.763	0.044
10. Furniture	0.182	0.103	0.060	0.690	0.023
11. Healthcare	-0.001	0.009	0.015	0.760	0.018
12. Transportation	0.560	0.329	0.202	1.270	0.078
13. Leisure and culture	-0.007	0.277	0.433	0.574	0.027

Indeed, Table 11 shows that rebound effects are highest for the first income decile and lowest for the tenth decile. According to our estimations, switching to a vegetarian diet would lead to backfire effects in the energy use of low-income consumers, and reuse nearly all the GHG savings initially incurred. On the other hand, the tenth decile saves both in energy use and greenhouse gas emissions even after re-spending the saved income. This is intuitive since they tend to re-spend primarily on luxury goods, which are less environmentally intensive, while consumers in the first decile tend to re-spend their income on necessities such as car use and household appliances.

Table 11: Comparison of rebound effects across income deciles

Rebound category	First decile	Average consumer	Tenth decile
Energy rebound effect	1.302	0.961	0.756
GHG rebound effect	0.879	0.494	0.253

These results confirm both the previous literature's observations that rebound effects tend to decline in size with increased income (Lenzen & Dey 2002; Murray 2013) as well as our research hypothesis (H4).

They also lead to interesting policy recommendations – from this vantage point, it would be most effective to encourage the highest income consumers to forego meat more often, whereas a similar encouragement of the households with the lowest income could lead to adverse effects, especially considering their energy footprints. Of course, the alternative policy avenue would be to refuse to accept this re-spending behavior as a foregone conclusion and instead advocate for environmentally 'smarter' ways of rebounding. Some of these alternatives are detailed in Section 7.6.3.

7.6. Sensitivity analyses

The results presented above rest on many assumptions. It is of great interest to test some of these assumptions and the sensitivity of this paper's conclusions to them. This section will do this by first changing the procedure of constructing the new vegetarian diet; second, by investigating other functional forms of the Engel curves; and lastly, by relaxing the assumption of fixed preferences to investigate alternative rebounding scenarios.

7.6.1. Alternative vegetarian diet

As an alternative to the previous scenario, one could construct a new vegetarian diet using all available information from Haddad and Tanzman's (2003) study and scaling all items according to their average values, independently of whether they had shown to be significantly different between the vegetarian and non-vegetarian survey respondents.

Note that the alternative (non-selective) scenario would construct the first-round expenditure as follows:

$$x_i^{First\ Round} = \begin{bmatrix} 0 & if\ i\ belongs\ to\ fish\ or\ meat\ category \\ \frac{cons_i^{Veg}}{cons_i^{Non-veg}} * x_i^{Original} & otherwise \end{bmatrix}$$

The main difference to the previous scenario was to scale down the consumption of bread, milk, fats, potatoes, French fries, sugar and sweets, while scaling up the values for alcoholic beverages, and select fruit and vegetable categories.

In this case, the new vegetarian energy use (before rebound effects) $E^{First\ Round}$ is 193 278 MJ/person/year, which is slightly more than the initial result of 193 097 MJ/person/year. $E^{Second\ Round}$, calculated with WLS double semi-log marginal expenditure shares, is 196 764 MJ/person/year in the case of an average consumer. Therefore, the calculated energy rebound effect is 102.8%, and we can see a moderate backfire effect from the new diet. In terms of greenhouse gas impacts, the new $E^{First\ Round}$ is 9 061 kg CO_2-eq/person/year, similarly slightly higher than the original 8 994 kg. With a new $E^{Second\ Round}$ of 9 257 kg CO_2-eq/person/year, the GHG rebound effect is 60.77%. This result is significantly higher than the baseline estimate of 49.39%. We can also compare the baseline and alternative dietary assumptions across income levels, as done in Table 12. In general, it seems that the different dietary assumptions matter more when calculating the rebound effect concerning greenhouse gas emissions than when investigating energy consumption. Yet, the differences are not important enough to change conclusions stemming from the present analysis.

Table 12: Comparison of rebound effects across vegetarian diet scenarios

Rebound effect	Baseline vegetarian diet	Alternative vegetarian diet
Energy use, first decile	1.302	1.480
Energy use, average consumer	0.961	1.028
Energy use, tenth decile	0.756	0.762
GHG emissions, first decile	0.879	1.184
GHG emissions, average consumer	0.494	0.608
GHG emissions, tenth decile	0.253	0.347

7.6.2. Alternative functional forms

As discussed in Section 4.3.3, one can choose from a variety of functional forms to estimate Engel curves and relatedly calculate marginal expenditure shares for different income categories. While the present study is based on using the Weighted Least Squares double semi-log estimation procedure due to theoretical consistency and the fit of estimated values, it is of interest to compare results of using alternative functional forms to test the sensitivity of the estimation to the choice of method. This has been done in Table 13.

Table 13: Comparison of rebound effects across functional forms

Model	First decile		Average		Tenth decile	
	Energy rebound	GHG rebound	Energy rebound	GHG rebound	Energy rebound	GHG rebound
OLS Linear	0.922	0.538	1.002	0.524	1.003	0.510
WLS Linear	0.876	0.501	0.952	0.487	0.953	0.474
OLS Quadratic	1.222	0.783	0.992	0.517	0.671	0.274
WLS Quadratic	1.207	0.803	0.983	0.513	0.669	0.249
OLS Working-Leser	0.959	0.534	1.040	0.556	1.104	0.499
WLS Working-Leser	0.928	0.508	0.989	0.517	1.037	0.458
OLS Double Semi-Log	1.312	0.851	0.967	0.499	0.761	0.273
WLS Double Semi-Log	1.302	0.879	0.961	0.494	0.756	0.253

It is apparent that the values for the average consumer do not differ much across estimation procedures: we seem to consistently estimate an energy rebound effect of around 96% to 100% and a GHG rebound effect of around 50%.

However, when estimating rebound effects for the first and tenth decile, respectively, we can differentiate between two groups of results. The linear and Working-Leser results predict that the first decile will have lower rebound effects than an average consumer – between 87% and 96% – in terms of energy use, and practically identical results regarding the GHG rebound – between 50 and 53%. Furthermore, the tenth decile is predicted to have practically identical energy use rebound effects ranging from 95% to 103%, and slightly lower GHG rebound effects between 46% and 51%. These results are intuitive since in these cases marginal expenditure shares are predicted to be identical (in the linear case) or roughly similar (in the case of Working-Leser), as can be seen in Section 6.2.1. Thus, the differences in rebound effect derive in first place from the different amounts of meat consumed and money re-spent.

The case is different for the quadratic and double semi-log estimation results. Here, the quadratic estimates confirm the baseline (double semi-log) results that predict a higher rebound effect for lower-income individuals – between 120% and 131% in energy use and from 78% to 87% in greenhouse gas emissions – and a lower effect for higher-income consumers – from 67% to 76% in energy use and between 25% and 27% regarding greenhouse gas footprints. The results therefore show that the functional form used – especially the amount of curvature allowed for the Engel curves of interest

– is of particular importance when estimating income-differentiated values. Yet, the fact that all values converge for the average consumer allows us to have considerable confidence in the baseline results.

7.6.3. Alternative re-spending scenarios

For the sake of policy analysis, it is of interest to relax the assumption of constant preferences of consumers and investigate different alternative re-spending patterns in order to advise households on the best way of making consumption pattern changes. This will be done for three scenarios: first, we follow Carlsson-Kanyama et al. (2005) in assuming that environmentally aware consumers switch to an all-organic diet as well as to vegetarianism; second, we take inspiration from Druckman et al. (2011) and model confining the re-spending of the saved expenditure to the least environmentally intensive category, and third, we repeat the analysis with the most environmentally intensive category in order to explore the scope of the rebound effect possible.

Organic food consumption

If a vegetarian diet is chosen for its environmental value, it is possible that consumers will simultaneously choose organic instead of conventional products for their perceived improved impact on the environment. Organic products however are often associated with a price premium, which can fundamentally change the conclusions on re-spending and environmental impacts occurred. In fact, a 2004 study by the Swedish Consumers' Agency (Konsumentverket) found that an average consumption basket was around 46% more expensive when all goods were bought from an organic brand than if they were conventional products (Konsumentverket 2004). Carlsson-Kanyama et al. (2005) similarly find price premiums of 30% to 50% in on-site estimations made in Stockholm.

Furthermore complicating the analysis, the production of organic products can be associated with improved environmental outcomes compared to conventional products (Taylor 2000). The EAP program used for our primary analysis predicted that on average, the energy intensity of organic produce was around 7% lower (Carlsson-Kanyama et al. 2005). Unfortunately, there was no information available on differences in greenhouse gas emissions, though in the case of produce that is likely to be closely connected to the embodied energy requirements. I thus assumed a similar 7% decrease in GHG-intensity values. Furthermore, for consistency, I followed Carlsson-Kanyama et al. (2005) in assuming that all fruit, vegetables, potatoes and milk were organically produced, and that the remainder of the consumption remained the same as in the baseline scenario.

Thus, I proceeded with the sensitivity analysis as follows:

(i) I kept $E^{Original}$ constant and adjusted $E^{First\ Round}$ by using the original expenditures, but 7% lower environmental intensities in the new organic categories. It is necessary to use the original expenditures because the model assumes constant prices and would thus associate the higher (organic) expenditures with larger quantities consumed, which would naturally skew the analysis. This leads me to first-round footprints of 192 793 MJ/person/year and 8 953 kg CO_2-equivalents/person/year.

(ii) I then multiplied all organic food expenditures in the first round scenario by 1.46 to model a 46% higher cost of the organic diet. This balances out the savings from foregoing meat and fish nearly perfectly, only leading to higher expenditures by 72 SEK/person/year, which in the second round scenario will be saved by reducing expenditure in other sectors according to the marginal expenditure shares.

(iii) This procedure leads to a new vegetarian expenditure pattern at 163 828 SEK which is associated with an energy use of 192 714 MJ/person/year and greenhouse gas emissions of 8 949 kg CO_2-equivalents/person/year. Therefore, not only would we avoid any rebound behavior, we find virtuous cycle effects in that consumers will save 111% of both the potential energy and the potential greenhouse gas emission savings that were predicted from only foregoing meat alone.

Carlsson-Kanyama et al. (2005) find that the new diet leads to a reduction in household energy use for food consumption of 13–32%. This compares favorably with our results, which show a reduction of the environmental footprint related to food and beverages by 18% (energy) to 22% (GHG).

These results show that advising consumers to switch toward both less meat-intensive and organic diets could have significantly higher environmental benefits than just focusing on one particular sector alone.

Best-case and worst-case re-spending scenarios

In addition to modeling the most likely re-expenditure behavior, Druckman et al. (2011) also consider the best-case and worst-case scenarios where all saved income is re-spent in the least or most environmentally intensive category respectively. We replicate this approach, using the mean intensities of the 13 higher-level categories. Table 14 shows these categories ordered by energy use and GHG emission intensity respectively.

Table 14: Lowest and highest environmental intensities

Categories ordered by energy intensity		Categories ordered by GHG intensity	
Category	MJ/SEK	Category	kg CO_2-eq/SEK
5. Tobacco	0.100	5. Tobacco	0.004
7. Services	0.367	7. Services	0.008
4. Alcoholic beverages	0.443	3. Restaurant visits	0.011
3. Restaurant visits	0.490	11. Healthcare	0.018
13. Leisure and culture	0.574	10. Furniture	0.023
2. Non-alcoholic beverages	0.607	8. Clothes and shoes	0.027
10. Furniture	0.690	13. Leisure and culture	0.027
8. Clothes and shoes	0.721	4. Alcoholic beverages	0.029
11. Healthcare	0.760	6. Consumables	0.030
1. Food	0.770	2. Non-alcoholic beverages	0.041
6. Consumables	0.983	9. Housing (rent, energy)	0.044
12. Transportation	1.270	12. Transportation	0.078
9. Housing (rent, energy)	1.763	1. Food	0.078

Note: '1. Food' category excluding meat and fish products to account for the vegetarianism restriction.

It can be seen that the two most benign categories overlap for energy and GHG intensity. From a public health perspective, it might be problematic to encourage a substantial increase in tobacco use, though. Therefore I chose re-spending in the 'Services' category as the best-case scenario. This category includes social services such as childcare and insurances, but also other services that are usually dematerialized – that is, they do not require large material inputs (Carlsson-Kanyama 1998). When re-spending all saved income on services, we arrive at new footprints of 194 235 MJ (symbolizing an energy rebound effect of 31.8%) and 9 018 kg CO_2-equivalent (which is a rebound of only 6.3%). On the other hand, when considering the worst-case scenarios, in terms of energy use we model re-expenditure exclusively on housing and reach rebound effects of 152.9% (in effect, constituting a backfire effect where the average consumer uses 198 562 MJ, 0.96% more energy than before the change). In terms of greenhouse gas emissions, we re-spend only on the food (or, alternatively, transportation) category and find rebound effects of 62%, with a new footprint of 9 235 kg CO_2-equivalent emissions.

Table 15: Comparison of rebound effects across re-spending scenarios

	Organic scenario	Best-case scenario	Worst-case scenario
Energy use after re-spending	192 714 MJ	194 235 MJ	198 562 MJ
Energy use rebound effect	- 11%	31.8%	152.9%
GHG emissions after re-spending	8 949 kg	9 018 kg	9 235 kg
GHG emissions rebound effect	- 11%	6.3%	62.0%

This sensitivity analysis shows that the particular rebound behavior really matters when considering the benefits of a behavior change. Saving money or spending it on dematerialized services can maximize benefits, while spending it on environmentally intensive goods would do more harm than good. This should be kept in mind when creating policy recommendations.

8. Conclusions and Recommendations

This final chapter first summarizes the insights resulting from the present model. It then turns to policy recommendations that can be derived from them. Finally, it points out limitations of the analysis, related to both data availability and methodological choices, discusses ethical considerations, and suggests topics for future research.

8.1. Synthesis of the results

The present study has shown that the correct consumption choices for sustainability depend on a large number of factors. Switching to a modeled vegetarian diet could save an average Swedish consumer as much as 16% of their energy use on food and beverages and even 20% of the greenhouse gas emissions related to their dietary consumption. This corresponds to driving between 2 500 and 4 000 kilometers a year. However, if Swedish consumers re-spend the money saved in their usual patterns, they would forego almost all of the potential energy savings and nearly half of the greenhouse gas emission savings, as we find rebound effects of 96% and 49% respectively. These rebound effects are even higher for lower-income consumers, since they tend to re-spend on more environmentally intensive goods, while we predict high-income consumers to have lower rebounds. As the sensitivity analysis shows, the adverse effect could be tempered by simultaneously purchasing organic goods or by re-spending the money exclusively on services.

8.2. Recommendations

The analysis has shown that it is insufficient for sustainable consumption policies to focus on consumption categories in isolation. Unless one considers the entire purchasing behavior of individuals, campaigns to convince consumers to change their choices one at a time may have no notable or even adverse effects. In general, rebound effects will always occur as long as the available income stays constant, although as we have seen it will decrease with the degree of dematerialization of the substitute purchases. The most sustainable recommendation to give consumers from this microeconomic perspective is to decrease their purchases of environmentally intensive goods – such as meat – and simultaneously to reallocate their expenditure to higher quality goods with a larger per-item price tag. Organic products are an example in the food category, but so are items produced according to other social and environmental standards such as Fair Trade or FSC. In the long run, the most sustainable way to reduce the footprint of individual consumption would be to convince consumers to explore maximizing their utility in non-material ways, for example by choosing a lower amount of working hours and exchanging more of their income for leisure time.

8.3. Limitations, ethics and future research

There are a number of limitations that need to be considered when interpreting and extrapolating from the results of the present study. They relate both to the data quality as well as to the methodologies used. Furthermore, we need to address the ethical aspects of both research methods and goals. This section will conclude by pointing out interesting areas of future research.

8.3.1. Limitations in data quality

Collecting data regarding household consumption behavior has many challenges attached to it. Both the household expenditure survey and the comparison of vegetarian and non-vegetarian diets are based on recall surveys, in which households are asked to detail their expenditures and consumption over a number of days. Extrapolating this information may misrepresent total annual consumption patterns if the data was collected during days unrepresentative of the household's average behavior. Furthermore, respondents tend to forget, over- or underestimate some items, which affects the reliability of the collected data (Reinders et al. 2003). Also, expenditures on household goods are often not continuous, which allows for a certain margin of error if the decision to acquire an item with relatively large costs attached to it – say, a fridge – was randomly made in the time period in question and could just as easily been put off until a later time. However, the use of semi-aggregated data alleviates this concern to some extent, as such random occurrences can be balanced out between different individual observations.

As mentioned, the household expenditure survey only had a response rate of 50%, which in effect means that the conclusions reached in this paper are representative only for the Swedish consumers that responded to the survey. It is possible that there exist socio-economic or other characteristics that make them more likely to respond than other consumers, or which are correlated with such reasons. This should be kept in mind when interpreting the results.

While the American consumption survey recorded items at a relatively high level of disaggregation, there were some items of interest – most importantly, eggs, but also tofu or other processed "faux" meats commonly used as vegetarian replacements – that were not separately logged and could thus not be adjusted. Similarly, the Swedish expenditure survey did not provide a separate category for tofu or legumes, forcing us to operate at a higher level of aggregation. This may have lead to estimation errors.

Furthermore, the assumption that American food consumption behavior – regarding the proportional difference between vegetarian and non-vegetarian lifestyles – can be used to model Swedish dietary habits may also be challenged. It would thus be helpful to investigate Swedish dietary patterns more closely in order to make the analysis even more consistent.

Then, the estimation of the environmental impact is based on the environmental intensity data provided by the Swedish Defense Research Agency, and the accuracy of the footprint estimates thus hinges on the accuracy of their findings. While these researchers carried out an elaborate process of calculating life cycle impacts for the different product categories, certain simplifying assumptions had to be made in both the aggregation of the different categories and the choice of goods analyzed that might skew the findings (Johansson et al. 2010). Furthermore, the use of per-crown intensity indicators might overestimate the true energy or greenhouse gas footprints of higher-income consumers because it assumes proportional quantity increases and does not account for quality differences that might be responsible for higher expenditures (Reinders et al. 2003; Murray 2013).

8.3.2. Limitations in estimation procedures

The use of Engel curves to model household consumption and rebound behavior, while efficient, is a simplification of reality that has a number of limitations. First, the focus on the income-expenditure relationship ignores possible price differences, socio-economic, environmental or geographical characteristics that may lead to different preferences and associated differences in consumption. This focus is strengthened in this study by the use of grouped means rather than individualized household data points. It thus needs to be clear that results are valid for a stylized model consumer and could be significantly different in reality.

Furthermore, the use of data collected over different time periods within a year, the method used by Statistics Sweden, introduces transitory and seasonal components into the analysis and can lead to an overestimation of the Engel elasticity for some luxury items and an underestimation of the elasticity of some necessary items (Haque 2005). It should also be noted that meals away from the home was coded as a separate category in which it was impossible to differentiate between vegetarian and non-vegetarian consumption. This category was thus excluded from the behavior change and rebound scenario, meaning that the estimated environmental savings represent a lower bound; the beneficial impact could be even greater if meat is also eschewed when eating out.

Finally, it should be restated that the temporal and theoretical scope of the research question is of major importance when considering the generalizability and the limitations of the present results. As previously explained, these estimations hold for a single average Swedish consumer with fixed preferences in the short run and markets in which prices stay fixed as well. Conclusions might be very different if considering long-run market adjustments (regarding prices, demand and supply), changes of preferences, adjustments of work-leisure decisions, and full-scale shifts of land use.

8.3.3. Ethical considerations

This study touches upon a multitude of ethical dimensions. First, the policy goal and research motivation itself – to improve the sustainability of current consumption patterns – arises from a perceived inter-temporal responsibility for the welfare of future generations. However, addressing this goal from the consumption side also involves a value judgment on the supremacy of collective welfare and satisfaction over individual utility and the freedom of choice. Advising consumers to change their behavior, or possibly even mandating it on an institutional level – such as the debated introduction of 'Meatless Mondays' in German workplace cafeterias in 2013 (Hawley 2013) – thus requires solid evidence of the societal benefit of such individual steps. This study hopes to provide policy-makers with more information in this regard.

Yet, the projection of average consumer behavior should never be interpreted as an accurate prediction of the choices of an individual consumer, as these projections are based on abstraction and simplifying assumptions. For example, there might well be low-income households with very low rebound effects, or high-income consumers that would experience backfire effects when switching to a vegetarian diet. Furthermore, the previous sections have outlined the study's limitations both in terms of data and of methodology. In order to prevent confusion and the possible spreading of misinformation, any policy steps undertaken based on studies as the one present should thus be very explicit in communicating the assumptions and methods used for the presented results. Otherwise, there could be a danger of abusing the trust consumers place in the political and scientific sector to help them tackle intricate problems such as the sustainability of their food consumption choices.

8.3.4. Future research

Despite the increased focus on sustainable food consumption by both political actors and civil society, the research regarding the environmental impacts and the related rebound effects of different dietary habits is only at its infancy. In further steps, it would be interesting to carry out a similar analysis for other suggested types of 'sustainable diets' – such as veganism or a shift toward the Mediterranean diet –, and to compare the benefits and drawbacks of different types of re-spending behavior further.

More research is also necessary on the real consumption patterns that consumers with different dietary self-definitions (such as vegetarian, vegan, gluten-free, etc.) follow. Furthermore, because the analysis relies so much on very local data, it would be insightful to compare impacts across regions and countries. Finally, more methodological work could be carried out in refining estimation procedures to model consumption choices, including improving the estimation of Engel curves and the comparison of Engel curves with other models.

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10. Appendices

10.1. Appendix 1: Expenditure categories and environmental intensities

	Average household expenditure	Energy intensity MJ/kr 2006	GHG intensity kg CO2-eq/kr 2006	Adjustment
Total expenditures	**163827.99**			
1. Purchased food	**19177.79**	**0.82**	**0.082**	
1.1.Bread, cereals	3198.20			
1.1.1. Rice and rice products	142.45			
1.1.1.1. Rice	67.15	0.95	0.057	
1.1.1.2. Rice products	75.31	0.95	0.057	
1.1.2. Pasta and pasta products	239.18	0.91	0.054	
1.1.3. Flour, grains	435.87	0.87	0.047	used "andra spannmålsprodukter"
1.1.4. Bread	1385.17			
1.1.4.1. Hard bread	204.33	0.86	0.040	
1.1.4.2. Soft white bread	303.55	0.86	0.044	
1.1.4.3. Soft dark bread	258.90	0.86	0.044	
1.1.4.4. Soft light bread	375.00	0.86	0.044	
1.1.4.5. Other bread	243.39	0.87	0.047	used "andra spannmålsprodukter"
1.1.5. Pastries	995.53	0.82	0.040	
1.2. Meat	3788.26			
1.2.1. Fresh, cooled or frozen beef	530.13	0.98	0.156	
1.2.2. Fresh, cooled or frozen pork	753.80	0.96	0.110	
1.2.3. Fresh, cooled or frozen other meat	81.46	0.73	0.120	
1.2.4. Fresh, cooled or frozen poultry	361.75	0.94	0.059	
1.2.5. Dried, salted or smoked beef	0.00	0.81	0.077	used "annat konserverat eller bearbetat kött"
1.2.6. Dried, salted or smoked pork	788.55	0.86	0.068	
1.2.7. Dried, salted or smoked other meat	177.75	0.81	0.077	used "annat konserverat eller bearbetat kött"
1.2.8. Sausage and sausage spreads	609.62	0.87	0.094	
1.2.9. Pate and charcuterie	192.13	0.81	0.077	used "annat konserverat eller bearbetat kött"
1.2.10. Meat dishes and other meat products	218.93	0.81	0.077	used "annat konserverat eller bearbetat kött"
1.2.11. Meat - other	74.13	0.81	0.077	used "annat konserverat eller bearbetat kött"

1.3. Fish and seafood	1183.66			
1.3.1. Fresh, cooled or frozen fish	418.07			
1.3.1.1. Cod fish	103.09	1.23	0.120	
1.3.1.2. Salmon	177.25	0.84	0.092	
1.3.1.3. Herring	25.84	1.38	0.079	
1.3.1.4. Other fish	111.89	1	0.083	used "annan konserverad eller beredd fisk"
1.3.2. Seafood	185.79	1.07	0.14	
1.3.3. Kaviar, roe, other fish products	99.01	1	0.083	used "annan konserverad eller beredd fisk"
1.3.4. Fish and seafood dishes	463.19	1	0.083	used "annan konserverad eller beredd fisk"
1.3.5. Fish and seafood - other	18.25	1	0.083	used "annan konserverad eller beredd fisk"
1.4. Milk, cheese and eggs	3130.16			
1.4.1. Whole milk (Fat >=1,5%)	189.99	1.18	0.220	
1.4.2. Low-fat and skim milk (Fat <1,5%)	568.37	1.18	0.220	used mjölk > 1,5 procent
1.4.3. Powdered and canned milk	0.00	0.93	0.200	used "andra mjölkprodukter"
1.4.4. Fil milk and yoghurt	528.66	0.955	0.137	used average of sweetened and unsweetened
1.4.5. Cream, sour cream, crème fraiche (Fat>=29%)	150.47	0.93	0.200	used "andra mjölkprodukter"
1.4.6. Cream, sour cream, crème fraiche (Fat<29%)	194.30	0.93	0.200	used "andra mjölkprodukter"
1.4.7. Cheese (Fat >=17%)	786.63	1.1	0.230	used hårdost
1.4.8. Cheese (Fat <17%)	453.20	1.1	0.230	used hårdost
1.4.9. Eggs	243.54	0.82	0.083	
1.4.10. Milk, cheese and eggs - other	16.30	0.93	0.200	used "andra mjölkprodukter"
1.5. Oils and fats	481.87			
1.5.1. Butter	156.05	1.38	0.250	
1.5.2. Margarine (fat>=40%)	132.21	1.45	0.029	
1.5.3. Margarine (fat<40%)	81.01	1.45	0.029	
1.5.4. Olive oil, table oil, mayonnaise	112.60	0.47	0.060	
1.6. Fruit and berries	1660.85			
1.6.1. Apples	217.89	0.38	0.048	
1.6.2. Pears	76.96	0.39	0.049	
1.6.3. Bananas	299.17	0.28	0.052	
1.6.4. Citrus fruit	264.28	0.35	0.036	
1.6.5. Other fresh fruit	214.55	0.35	0.046	used average of "äpplen", "päron", "bananer" and "citrusfrukter"
1.6.6. Dried fruit and berries, nuts	273.05	0.77	0.040	
1.6.7. Berries	220.24	0.24	0.024	
1.6.8. Fruit and berry conserves	54.59	0.43	0.037	used average of "äpplen",

				"päron", "bananer" and "citrusfrukter"
1.6.9. Fruit and berries - other	40.11	0.45	0.037	used average of "äpplen", "päron", "bananer" and "citrusfrukter"
1.7. Vegetables	2126.09			
1.7.1. Salad, different varieties, fresh	255.18	0.31	0.048	used average of "kål, färsk" and "lök, purjolök, färsk"
1.7.2. Cabbage, fresh	80.03	0.26	0.033	
1.7.3. Tomatoes, peppers, etc., fresh	724.71	0.91	0.100	
1.7.4. Onion, green onion, fresh	116.44	0.36	0.063	
1.7.5. Mushrooms, fresh	38.04	0.41	0.023	
1.7.6. Soups, salads, vegetable dishes	326.53	0.44	0.047	used average of "kål", "grönsaker som odlas", "lök", "svamp", "rotfrukter", "potatis"
1.7.7. Root vegetables	115.15	0.26	0.026	
1.7.8. Potatoes	216.01	0.41	0.036	
1.7.9. Potato products (french fries, instant mashed potatoes etc.)	210.10	0.83	0.044	
1.7.10. Vegetables - other	43.91	0.44	0.047	used average of "kål", "grönsaker som odlas", "lök", "svamp", "rotfrukter", "potatis"
1.8. Sweets, sugar	1794.96			
1.8.1. Sugar, syrup, honey, sweeteners	123.92	0.80	0.045	
1.8.2. Jam, marmelade	104.39	0.84	0.067	
1.8.3. Ice cream	360.09	1.60	0.120	
1.8.4. Other sweets, candy, chocolate	1206.57	0.78	0.031	
1.9. Sauces, dressings, condiments	322.71	0.91	0.058	
1.10. Salt and spices	194.40	0.79	0.026	
1.11. Baking powder, bouillon, etc.	173.16	0.79	0.026	used "salt, kryddor och kryddväxter"
1.12. Snacks	210.73	0.78	0.036	
1.13. Other food	912.73	0.82	0.082	used average of all other livsmedel categories
2. Non-alcoholic beverages	**1578.52**	**0.61**	**0.041**	
2.1. Fruit and vegetable juice, nectar	290.17	0.14	0.035	
2.2. Sodas	624.59	1.10	0.067	
2.3. Mineral water	126.80	1.10	0.050	
2.4. Coffee	418.26	0.05	0.040	
2.5. Tea	92.00	0.57	0.026	
2.6. Cocoa	26.69	0.68	0.026	
3. Restaurant meals	**6460.43**	**0.49**	**0.011**	used "restauranger, kafeer och liknande"
4. Alcoholic beverages	**2247.07**	**0.44**	**0.029**	#average of "spritdrycker, vin

				and lättöl"
5. Tobacco	**1170.46**	**0.10**	**0.004**	
6. Consumables	**3502.21**	**0.98**	**0.030**	
6.1. Personal hygiene	2082.33	1.10	0.025	used "artiklar för rengöring"
6.2. Other consumables	1419.88	0.87	0.034	used average of "hushållspapper", "toalettpapper", and "artiklar för rengöring"
7. Household services	**7810.98**	**0.37**	**0.008**	
7.1. Childcare	968.98	0.22	0.003	used "daghem och lekskolor"
7.2. Labor union fees, other insurance	3146.93	0.39	0.010	used "a-kassa och fackföreningsavgift"
7.3. Other services	3695.07	0.49	0.012	used "frisersalonger och skönhetssalonger"
8. Clothes and shoes	**8025.21**	**0.72**	**0.027**	
8.1. Clothes	6582.04			
8.1.1. Outdoor clothes	1483.70	0.68	0.019	used "regnkläder"
8.1.2. Other clothes	3986.27	0.68	0.019	used average of "klänningar av tyg" and "kläder, ospec. Av päls eller skin"
8.1.3. Underwear	859.11	0.77	0.023	
8.1.4. Accessories	252.97	0.76	0.022	
8.2. Shoes	1443.17	0.72	0.052	
9. Housing	**38878.43**	**1.76**	**0.044**	
9.1. Rent (incl. garage)	18970.57	1.2	0.026	
9.2. Repairs	3057.93	0.31	0.008	used "reparationer av möbler och golvbeläggningar"
9.3. Home and house insurance	1223.14	0.35	0.005	used "bilförsäkring"
9.4. Housing services	2382.55	0.48	0.011	used "vattenförsörjning och diverse andra tjänster förknippade med bostaden"
9.5. Electricity, gas and other fuels	5882.55	7.04	0.186	used average of "elektricitet", "gas", "flytande bränslen", "fasta bränslen", "värmeenergi" (potential source of limitation since not disaggregated)
9.6. Mortgages (brutto)	7361.69	1.20	0.026	
10. Furniture, inventory, textiles, household appliances	**9325.91**	**0.69**	**0.023**	
10.1. Furniture and inventory, mats and other floor covers	4485.36	0.66	0.020	used average of "möbler av trä", "persienn" and "matta"
10.2. Textiles	994.68	0.70	0.019	used "påslakan, örngott, underlaken"
10.3. Household appliances	3845.87	0.71	0.028	used average of "kylskåp", "diskmaskin", "tvättmaskin", "torkskåp", and "brödrost"
11. Healthcare	**3784.08**	**0.76**	**0.018**	used average of "farmaceutiska produkter", "vitaminer, hälsokost", "andra medicinska produkter", "glasögon"

12. Transportation	**30796.60**	**1.27**	**0.078**	
12.1. Car purchase	12439.99	0.68	0.021	
12.2. Purchase of other personal vehicle	0.00	0.62	0.021	used average of "motorcykel, moped, skoter" and "cyklar"
12.3. Car use	13801.03	3.25	0.235	used average of "bensin till bil" and "diesel till bil"
12.4. Loan payment (car - brutto), car tax	1199.60	0.15	0.004	used "finansiella tjänster"
12.5. Use of other personal vehicle	747.68	3.25	0.235	used average of "bensin till bil" and "diesel till bil"
12.6. Loan payment (not car - brutto), vehicle tax (not car)	0.00	0.15	0.004	used "finansiella tjänster"
12.7. Local trips, public transportation	2608.29	0.80	0.029	used average of "tågbillett" and "busbiljett"
13. Leisure and culture	**31070.31**	**0.57**	**0.027**	
13.1. Holiday home	1713.18	0.55	0.013	used "hotell"
13.2. Radio and tv	2006.47	0.60	0.026	used average of "radio" and "TV"
13.3. Games, sports, hobby	3374.51	1.10	0.040	
13.4. Camera, photo services	1012.69	0.45	0.015	
13.5. Travel, hotel	7263.75	0.83	0.059	used average of "inrikesresa", "utrikesresa" and "hotell'
13.6. Other leisure	5586.54	0.44	0.070	used average of "kamera", "båt", trädgårdsväxter", "husdjur" and "museer"
13.7. Entertainment	736.39	0.46	0.009	used "bio"
13.8. Books, newspapers, TV-licence, etc.	5620.13	0.62	0.023	
13.9. Mobile phone (Calls or plan)	1491.84	0.35	0.008	
13.10. Fixed phone (Calls or plan)	2264.79	0.35	0.008	

10.2. Appendix 2: Marginal expenditure share results

	Average expenditure shares	Linear OLS	Linear WLS	Quadratic OLS	Quadratic WLS	Working-Leser OLS	Working-Leser WLS	Double Semi-Log OLS	Double Semi-Log WLS
Total expenditures	1	1.00164	1.00016	1.00014	1.00013	1.00010	1.00012	1.00039	1.00015
1. Purchased food									
1.1.Bread, cereals									
1.1.1. Rice and rice products									
1.1.1.1. Rice	0.00041	0.00020	0.00012	0.00018	0.00020	0.00024	0.00020	0.00014	0.00014
1.1.1.2. Rice products	0.00046	-0.00020	-0.00024	-0.00021	-0.00022	-0.00019	-0.00022	-0.00023	-0.00024
1.1.2. Pasta and pasta products	0.00146	0.00259	0.00128	0.00112	0.00110	0.00089	0.00108	0.00123	0.00124
1.1.3. Flour, grains	0.00266	0.00259	0.00232	0.00254	0.00257	0.00278	0.00258	0.00242	0.00238
1.1.4. Bread									
1.1.4.1. Hard bread	0.00125	0.00057	0.00054	0.00056	0.00055	0.00057	0.00055	0.00054	0.00054
1.1.4.2. Soft white bread	0.00185	0.00202	0.00196	0.00200	0.00208	0.00206	0.00206	0.00198	0.00199
1.1.4.3. Soft dark bread	0.00158	0.00057	0.00068	0.00059	0.00065	0.00038	0.00061	0.00067	0.00067
1.1.4.4. Soft light bread	0.00229	0.00184	0.00194	0.00186	0.00190	0.00167	0.00187	0.00195	0.00192
1.1.4.5. Other bread	0.00149	-0.00101	-0.00090	-0.00099	-0.00092	-0.00126	-0.00098	-0.00092	-0.00091
1.1.5. Pastries	0.00608	0.00207	0.00223	0.00208	0.00215	0.00179	0.00208	0.00213	0.00221
1.2. Meat									
1.2.1. Fresh, cooled or frozen beef	0.00324	0.00817	0.00852	0.00823	0.00843	0.00812	0.00840	0.00841	0.00849
1.2.2. Fresh, cooled or frozen pork	0.00460	0.00288	0.00243	0.00280	0.00251	0.00333	0.00262	0.00251	0.00247
1.2.3. Fresh, cooled or frozen other meat	0.00050	0.00170	0.00166	0.00170	0.00165	0.00177	0.00169	0.00170	0.00166
1.2.4. Fresh, cooled or frozen poultry	0.00221	0.00266	0.00305	0.00274	0.00280	0.00233	0.00276	0.00296	0.00298
1.2.5. Dried, salted or smoked beef	0.00000				0.00000	0.00000	0.00000		
1.2.6. Dried, salted or smoked pork	0.00481	0.00396	0.00404	0.00398	0.00413	0.00374	0.00406	0.00407	0.00406
1.2.7. Dried, salted or smoked other meat	0.00108	0.00147	0.00149	0.00145	0.00165	0.00153	0.00160	0.00142	0.00152
1.2.8. Sausage and sausage spreads	0.00372	0.00063	0.00065	0.00063	0.00067	0.00046	0.00061	0.00065	0.00065
1.2.9. Pate and charcuterie	0.00117	0.00069	0.00082	0.00072	0.00062	0.00056	0.00063	0.00079	0.00078
1.2.10. Meat dishes and other meat products	0.00134	-0.00019	-0.00040	-0.00024	-0.00031	-0.00004	-0.00028	-0.00037	-0.00037
1.2.11. Meat - other	0.00045	0.00047	0.00050	0.00048	0.00042	0.00046	0.00044	0.00049	0.00049
1.3. Fish and seafood									
1.3.1. Fresh, cooled or frozen fish									
1.3.1.1. Cod fish	0.00063	0.00008	0.00012	0.00008	0.00012	0.00002	0.00010	0.00010	0.00012
1.3.1.2. Salmon	0.00108	0.00116	0.00125	0.00118	0.00122	0.00104	0.00120	0.00125	0.00124
1.3.1.3. Herring	0.00016	-0.00070	-0.00045	-0.00065	-0.00062	-0.00099	-0.00066	-0.00051	-0.00050
1.3.1.4. Other fish	0.00068	0.00038	0.00042	0.00039	0.00037	0.00029	0.00037	0.00044	0.00041
1.3.2. Seafood	0.00113	0.00215	0.00178	0.00208	0.00197	0.00255	0.00204	0.00187	0.00183

1.3.3. Kaviar, roe, other fish products	0.00060	0.00077	0.00075	0.00077	0.00077	0.00081	0.00078	0.00075	0.00075
1.3.4. Fish and seafood dishes	0.00283	0.00143	0.00085	0.00129	0.00130	0.00198	0.00133	0.00090	0.00097
1.3.5. Fish and seafood - other	0.00011	0.00045	0.00048	0.00045	0.00051	0.00046	0.00049	0.00046	0.00048
1.4. Milk, cheese and eggs									
1.4.1. Whole milk (Fat >=1,5%)	0.00116	-0.00053	-0.00068	-0.00056	-0.00060	-0.00048	-0.00060	-0.00064	-0.00065
1.4.2. Low-fat and skim milk (Fat <1,5%)	0.00347	0.00219	0.00210	0.00217	0.00224	0.00214	0.00220	0.00215	0.00213
1.4.3. Powdered and canned milk	0.00000				0.00000	0.00000	0.00000		
1.4.4. Fil milk and yoghurt	0.00323	0.00363	0.00380	0.00367	0.00368	0.00346	0.00366	0.00378	0.00376
1.4.5. Cream, sour cream, crème fraiche (Fat>=29%)	0.00092	0.00126	0.00097	0.00120	0.00120	0.00155	0.00123	0.00102	0.00103
1.4.6. Cream, sour cream, crème fraiche (Fat<29%)	0.00119	0.00117	0.00118	0.00118	0.00115	0.00112	0.00116	0.00121	0.00117
1.4.7. Cheese (Fat >=17%)	0.00480	0.00204	0.00154	0.00196	0.00174	0.00239	0.00182	0.00168	0.00160
1.4.8. Cheese (Fat <17%)	0.00277	0.00308	0.00310	0.00308	0.00318	0.00309	0.00315	0.00308	0.00312
1.4.9. Eggs	0.00149	0.00048	0.00031	0.00045	0.00045	0.00059	0.00045	0.00034	0.00034
1.4.10. Milk, cheese and eggs - other	0.00010	-0.00031	-0.00027	-0.00030	-0.00030	-0.00036	-0.00031	-0.00028	-0.00028
1.5. Oils and fats									
1.5.1. Butter	0.00095	0.00017	0.00021	0.00018	0.00020	0.00011	0.00019	0.00041	0.00021
1.5.2. Margarine (fat>=40%)	0.00081	0.00001	-0.00018	-0.00003	-0.00008	0.00016	-0.00006	-0.00014	-0.00015
1.5.3. Margarine (fat<40%)	0.00049	0.00021	0.00011	0.00019	0.00016	0.00028	0.00017	0.00014	0.00012
1.5.4. Olive oil, table oil, mayonnaise	0.00069	0.00015	0.00025	0.00017	0.00016	0.00002	0.00015	0.00023	0.00023
1.6. Fruit and berries									
1.6.1. Apples	0.00133	0.00142	0.00129	0.00140	0.00139	0.00154	0.00140	0.00133	0.00131
1.6.2. Pears	0.00047	-0.00039	-0.00019	-0.00034	-0.00033	-0.00064	-0.00037	-0.00022	-0.00022
1.6.3. Bananas	0.00183	0.00099	0.00078	0.00095	0.00091	0.00113	0.00093	0.00083	0.00082
1.6.4. Citrus fruit	0.00161	0.00121	0.00102	0.00118	0.00110	0.00134	0.00113	0.00109	0.00104
1.6.5. Other fresh fruit	0.00131	0.00075	0.00112	0.00083	0.00087	0.00035	0.00082	0.00106	0.00106
1.6.6. Dried fruit and berries, nuts	0.00167	0.00145	0.00143	0.00144	0.00147	0.00146	0.00145	0.00143	0.00144
1.6.7. Berries	0.00134	0.00158	0.00144	0.00154	0.00159	0.00174	0.00159	0.00144	0.00148
1.6.8. Fruit and berry conserves	0.00033	0.00025	0.00021	0.00025	0.00024	0.00027	0.00024	0.00023	0.00022
1.6.9. Fruit and berries - other	0.00024	-0.00004	0.00004	-0.00003	-0.00003	-0.00012	-0.00004	0.00001	0.00002
1.7. Vegetables									
1.7.1. Salad, different varieties, fresh	0.00156	0.00203	0.00213	0.00206	0.00208	0.00193	0.00207	0.00214	0.00212
1.7.2. Cabbage, fresh	0.00049	0.00021	0.00019	0.00020	0.00019	0.00022	0.00019	0.00019	0.00019
1.7.3. Tomatoes, peppers, etc., fresh	0.00442	0.00517	0.00542	0.00523	0.00533	0.00489	0.00528	0.00541	0.00540
1.7.4. Onion, green onion, fresh	0.00071	0.00059	0.00059	0.00059	0.00063	0.00059	0.00061	0.00058	0.00060
1.7.5. Mushrooms, fresh	0.00023	0.00062	0.00069	0.00064	0.00063	0.00059	0.00063	0.00067	0.00067
1.7.6. Soups, salads, vegetable dishes	0.00199	0.00121	0.00110	0.00118	0.00113	0.00129	0.00115	0.00111	0.00111
1.7.7. Root vegetables	0.00070	0.00025	0.00024	0.00025	0.00026	0.00024	0.00025	0.00025	0.00024
1.7.8. Potatoes	0.00132	0.00129	0.00111	0.00126	0.00126	0.00146	0.00127	0.00115	0.00114

1.7.9. Potato products (french fries, instant mashed potatoes etc.)	0.00128	-0.00011	-0.00021	-0.00014	-0.00003	-0.00007	-0.00007	-0.00022	-0.00017
1.7.10. Vegetables - other	0.00027	0.00073	0.00078	0.00074	0.00074	0.00071	0.00075	0.00076	0.00077
1.8. Sweets, sugar									
1.8.1. Sugar, syrup, honey, sweeteners	0.00076	-0.00047	-0.00060	-0.00049	-0.00055	-0.00041	-0.00054	-0.00056	-0.00058
1.8.2. Jam, marmelade	0.00064	0.00041	0.00024	0.00038	0.00033	0.00058	0.00036	0.00028	0.00027
1.8.3. Ice cream	0.00220	0.00270	0.00245	0.00265	0.00266	0.00296	0.00268	0.00250	0.00251
1.8.4. Other sweets, candy, chocolate	0.00736	0.00414	0.00358	0.00402	0.00412	0.00442	0.00409	0.00374	0.00372
1.9. Sauces, dressings, condiments	0.00197	0.00159	0.00176	0.00162	0.00174	0.00143	0.00168	0.00170	0.00175
1.10. Salt and spices	0.00119	0.00141	0.00124	0.00138	0.00135	0.00156	0.00138	0.00129	0.00127
1.11. Baking powder, bouillon, etc.	0.00106	0.00054	0.00059	0.00055	0.00051	0.00047	0.00051	0.00058	0.00057
1.12. Snacks	0.00129	0.00138	0.00134	0.00136	0.00145	0.00140	0.00142	0.00134	0.00136
1.13. Other food	0.00557	0.00590	0.00650	0.00606	0.00592	0.00532	0.00591	0.00647	0.00636
2. Non-alcoholic beverages									
2.1. Fruit and vegetable juice, nectar	0.00177	0.00162	0.00181	0.00166	0.00173	0.00139	0.00169	0.00179	0.00178
2.2. Sodas	0.00381	0.00029	0.00052	0.00034	0.00042	-0.00017	0.00032	0.00051	0.00049
2.3. Mineral water	0.00077	0.00012	0.00039	0.00018	0.00016	-0.00015	0.00014	0.00034	0.00033
2.4. Coffee	0.00255	0.00072	0.00022	0.00062	0.00053	0.00110	0.00058	0.00033	0.00031
2.5. Tea	0.00056	0.00035	0.00034	0.00034	0.00037	0.00035	0.00036	0.00033	0.00035
2.6. Cocoa	0.00016	-0.00023	-0.00013	-0.00021	-0.00018	-0.00036	-0.00020	-0.00015	-0.00014
3. Restaurant meals	0.03943	0.06748	0.06844	0.06747	0.06868	0.06842	0.06864	0.06767	0.06845
4. Alcoholic beverages	0.01372	0.00990	0.00853	0.00947	0.01026	0.01138	0.01016	0.00838	0.00895
5. Tobacco	0.00714	-0.00631	-0.00787	-0.00659	-0.00710	-0.00556	-0.00703	-0.00744	-0.00765
6. Consumables									
6.1. Personal hygiene	0.01271	0.00904	0.00992	0.00914	0.00992	0.00820	0.00959	0.00952	0.00988
6.2. Other consumables	0.00867	0.01061	0.00951	0.01039	0.01042	0.01163	0.01052	0.00980	0.00975
7. Household services									
7.1. Childcare	0.00591	0.02229	0.02145	0.02211	0.02298	0.02356	0.02296	0.02179	0.02180
7.2. Labor union fees, other insurance	0.01921	0.04349	0.03945	0.04265	0.04306	0.04835	0.04359	0.04039	0.04036
7.3. Other services	0.02255	0.03678	0.03302	0.03596	0.03482	0.04160	0.03575	0.03347	0.03356
8. Clothes and shoes									
8.1. Clothes									
8.1.1. Outdoor clothes	0.00906	0.02255	0.02293	0.02256	0.02334	0.02301	0.02326	0.02269	0.02299
8.1.2. Other clothes	0.02433	0.04029	0.04199	0.04064	0.04122	0.03922	0.04107	0.04175	0.04176
8.1.3. Underwear	0.00524	0.00323	0.00344	0.00325	0.00356	0.00293	0.00342	0.00335	0.00345
8.1.4. Accessories	0.00154	0.00373	0.00337	0.00365	0.00366	0.00421	0.00372	0.00343	0.00344
8.2. Shoes	0.00881	0.01968	0.02353	0.02049	0.02080	0.01654	0.02056	0.02275	0.02282
9. Housing									
9.1. Rent (incl. garage)	0.11579	-0.32150	-0.30712	-0.31940	-0.32475	-0.35023	-0.32964	-0.31505	-0.31132
9.2. Repairs	0.01867	0.04657	0.04245	0.04571	0.04652	0.05149	0.04698	0.04349	0.04345
9.3. Home and house	0.00747	0.00557	0.00573	0.00563	0.00552	0.00526	0.00552	0.00577	0.00568

insurance									
9.4. Housing services	0.01454	0.02325	0.02123	0.02289	0.02326	0.02508	0.02342	0.02200	0.02173
9.5. Electricity, gas and other fuels	0.03591	0.04085	0.03843	0.04051	0.03961	0.04300	0.04015	0.03941	0.03878
9.6. Mortgages (brutto)	0.04493	0.13135	0.13357	0.13193	0.13357	0.13239	0.13393	0.13391	0.13347
10. Furniture, inventory, textiles, household appliances									
10.1. Furniture and inventory, mats and other floor covers	0.02738	0.05000	0.04591	0.04924	0.04882	0.05462	0.04958	0.04710	0.04668
10.2. Textiles	0.00607	0.01291	0.01024	0.01238	0.01234	0.01553	0.01264	0.01095	0.01078
10.3. Household appliances	0.02347	0.04303	0.04270	0.04305	0.04312	0.04391	0.04332	0.04314	0.04279
11. Healthcare	0.02310	0.00758	0.00926	0.00770	0.00874	0.00605	0.00815	0.00812	0.00908
12. Transportation									
12.1. Car purchase	0.07593	0.21015	0.21185	0.21063	0.21227	0.21409	0.21326	0.21233	0.21185
12.2. Purchase of other personal vehicle	0.00000				0.00000	0.00000	0.00000	0.00000	0.00000
12.3. Car use	0.08424	0.12856	0.10703	0.12439	0.12386	0.14915	0.12618	0.11302	0.11137
12.4. Loan payment (car - brutto), car tax	0.00732	0.00925	0.00672	0.00875	0.00860	0.01157	0.00887	0.00737	0.00721
12.5. Use of other personal vehicle	0.00456	0.00594	0.00458	0.00572	0.00516	0.00723	0.00547	0.00501	0.00476
12.6. Loan payment (not car - brutto), vehicle tax (not car)	0.00000					0.00000	0.00000	0.00000	0.00000
12.7. Local trips, public transportation	0.01592	-0.01840	-0.01405	-0.01762	-0.01761	-0.02339	-0.01832	-0.01558	-0.01495
13. Leisure and culture									
13.1. Holiday home	0.01046	0.03058	0.03458	0.03158	0.03071	0.02737	0.03085	0.03423	0.03362
13.2. Radio and tv	0.01225	0.01814	0.01767	0.01810	0.01718	0.01898	0.01761	0.01782	0.01760
13.3. Games, sports, hobby	0.02060	0.04158	0.04762	0.04288	0.04287	0.03686	0.04263	0.04647	0.04640
13.4. Camera, photo services	0.00618	0.00582	0.00449	0.00559	0.00577	0.00656	0.00575	0.00508	0.00481
13.5. Travel, hotel	0.04434	0.09044	0.10553	0.09377	0.09450	0.07753	0.09349	0.10316	0.10263
13.6. Other leisure	0.03410	0.05850	0.05911	0.05892	0.05644	0.05883	0.05747	0.05963	0.05856
13.7. Entertainment	0.00449	0.00597	0.00745	0.00627	0.00667	0.00455	0.00646	0.00717	0.00723
13.8. Books, newspapers, TV-licence, etc.	0.03430	0.00477	0.00243	0.00419	0.00489	0.00545	0.00450	0.00277	0.00303
13.9. Mobile phone (Calls or plan)	0.00911	0.00010	0.00104	0.00026	0.00038	-0.00107	0.00016	0.00070	0.00086
13.10. Fixed phone (Calls or plan)	0.01382	-0.00473	-0.00576	-0.00495	-0.00541	-0.00444	-0.00541	-0.00567	-0.00564