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The environmental pressures of foods

Application in climate taxation and sustainability assessment of diets

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Abstract

Reducing environmental impacts from the food system will require a shift towards environmentally sustainable eating patterns. To achieve changes in diets that reduce environmental burdens, a climate tax on food has been suggested. This thesis examined different aspects of data on environmental pressures and their use for establishing and evaluating a tax on food, as well as assessing the sustainability of diets. This included establishing climate impact values for foods, for use in a climate tax. Further, this thesis identified aspects needed to determine the environmental sustainability of diets in the Swedish context. The environmental pressures of the current average Swedish diet were calculated and benchmarked against suggested global environmental boundaries. Potential goal conflicts resulting from taxation were identified.

A method based on Life Cycle Assessment was developed for establishing consistent and transparent datasets on the climate impact of foods, for use in a climate tax on food. Evaluation of methodological choices for assessing climate impact revealed a common trade-off between using climate impact data that result in a theoretically cost-efficient tax and simplicity in calculations.

Comparison of the global EAT-*Lancet* framework for environmentally sustainable food systems and the national Swedish Environmental Objectives revealed a need for additional aspects to capture regional environmental concerns in Sweden. For this, there is a need for better inventory data, site-dependent impact modelling and improved traceability for imported foods. The environmental pressures of Swedish food consumption were found to exceed global boundaries for greenhouse gas emissions, cropland use and nutrient application by two- to four-fold. For extinction rate of terrestrial species, the boundary was transgressed by six-fold. The only environmental category for which the global boundary was not exceeded was freshwater use, for which the diet performed well below the limit.

Climate taxation on all foods on the Swedish market was found to have the potential to reduce food-related environmental pressures by 7-12%, mainly owing to an overall decrease in food consumption. With a decline in beef consumption, pasture use was found to decrease by up to 12%. To avoid potential goal conflicts with maintaining Swedish semi-natural pastures when introducing a climate tax, farmers could be given higher payments for management of these areas.

Keywords: Food, climate tax, environmental pressures, environmental impacts, environmental sustainability, Life Cycle Assessment, goal conflicts, EAT-*Lancet*, Swedish Environmental Objectives

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Livsmedels miljöavtryck - Tillämpning i klimatbeskattning och utvärdering av kosters hållbarhet

Sammanfattning

För att minska miljöpåverkan från livsmedelssystemet behövs en förändring av våra kostmönster. I denna avhandling undersöktes olika aspekter av hur en klimatskatt på mat skulle kunna påskynda en sådan förändring. Arbetet inkluderade att beräkna miljöavtryck för genomsnittliga livsmedel på den svenska marknaden för användning i framtagandet och utvärderingen av en skatt på mat, och för utvärdering av kosters miljömässiga hållbarhet. Detta innebar bland annat att undersöka hur data över livsmedels klimatpåverkan kan tas fram för att användas i en klimatskatt, och att identifiera luckor för att fånga den miljömässiga hållbarheten av kosten på lokal nivå i Sverige. I avhandlingen beräknades miljöavtrycket från den genomsnittliga svenska kosten och utvärderades mot globala gränser som föreslagits för livsmedelssystemet. Vidare ingick en identifiering av potentiella miljömässiga målkonflikter från beskattning på mat.

I avhandlingen utvecklades en metod baserad på livscykelanalys för att med en konsekvent metodik ta fram transparent data på livsmedels klimatpåverkan för användning i en klimatskatt på mat. Olika metodval för att beräkna klimatpåverkan utvärderades där resultaten från avhandlingen visar att valen ofta är en avvägning mellan att uppnå enkelhet i beräkningarna, och att ta fram data som resulterar i en teoretiskt kostnadseffektiv skatt.

I jämförelsen av det svenska miljömålssystemet och det globala EAT-*Lancet*-ramverket visar avhandlingen att ytterligare aspekter behöver lyftas in för att för att täcka in hållbarhetsaspekter av kosten på en lokal nivå. Detta kräver bättre indata, platsberoende modellering av miljöpåverkan, samt bättre statistik för att kartlägga produkters ursprungsländer.

Avhandlingens resultat visar att miljöavtrycken från svensk livsmedelskonsumtion ligger två till fyra gånger över de planetära gränserna för växthusgasutsläpp, användning av åkermark och näringstillförsel. För påverkan på den biologiska mångfalden leder nuvarande kostmönster till att den tillåtna gränsen överskrids med det sexdubbla. Den totala vattenanvändningen från kosten håller sig däremot långt under den uppskattade gränsen.

Resultaten i avhandlingen visar även att en klimatskatt på mat har potential att minska livsmedelskonsumtionens miljöavtryck med mellan 7 och 12%, vilket framförallt är en effekt av en minskad konsumtion av mat. På grund av minskningen i konsumtionen av nötkött minskade användningen av betesmark med upp till 12%. För att undvika en potentiell målkonflikt där en klimatskatt slår mot bete av naturbetesmark i Sverige skulle ersättningarna till naturbetesmarkerna kunna höjas vid införandet av en klimatskatt.

Nyckelord: Livsmedel, klimatskatt, miljöavtryck, miljöpåverkan, miljömässig hållbarhet, Livscykelanalys, målkonflikter, EAT-*Lancet*, Svenska miljömålen

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Dedication

To Inez

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I Gren, I.-M.*, Moberg, E., Säll, S. & Röös, E. 2019. Design of a climate tax on food consumption: Examples of tomatoes and beef in Sweden. *Journal of Cleaner Production* 211, 1576-1585.

II Moberg, E.*, Andersson, M.W., Säll, S., Hansson, P.-A. & Röös, E. 2019. Determining the climate impact of food for use in a climate tax—design of a consistent and transparent model. *The International Journal of Life Cycle Assessment* 24, 1715-1728.

III Moberg, E.*, Karlsson Potter, H., Wood, A., Hansson, P.-A. & Röös, E. 2020. Benchmarking the Swedish diet relative to global and national environmental targets—identification of indicator limitations and data gaps. *Sustainability* 12, 1407.

IV Moberg, E.*, Säll, S. Hansson, P.-A. & Röös, E. 2021. Environmental effects on a climate tax of food—identification of synergies and goal conflicts. *Food Policy* 101, 102090.

*Corresponding author.

The contribution of Emma Moberg to the papers included in this thesis was as follows:

I Planned the paper together with the co-authors. Performed the data collection and calculations of the climate impact. Assisted in calculating climate tax levels based on the results. Contributed to interpretation of the results. Assisted in writing the paper together with the co-authors.

II Planned the paper together with the co-authors. Carried out the data collection, calculations and interpretation of the results. Wrote the paper with input from the co-authors.

III Planned the paper together with the co-authors. Carried out the data collection, calculations and interpretation of the results. Wrote the paper with input from the co-authors.

IV Planned the paper together with the co-authors. Carried out the data collection, calculations and interpretation of the results. Wrote the paper with input from the co-authors.

Abbreviations

CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalents
dLUC	Direct land use change
E/MSY	Extinctions per million species-years
EPD	Environmental Performance Declaration
EU ETS	European Union Emissions Trading System
GHG	Greenhouse gas
GTP	Global temperature potential
GWP	Global warming potential
ICBM	Introductory Carbon Balance Model
iLUC	Indirect land use change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LUC	Land use change
N	Nitrogen
NH_3	Ammonia
NO_x	Nitrogen oxides
N_2O	Nitrous oxide
P	Phosphorus
PCR	Product Category Rule
PEF	Product Environmental Footprint
SEO	Swedish Environmental Objectives
VAT	Value-added tax

1 Introduction

The global food system is a major driver of many environmental pressures, including climate and land-system change, eutrophication and acidification of terrestrial and aquatic ecosystems, appropriation of freshwater resources, chemical pollution and biodiversity loss (Willett *et al.*, 2019). To reduce these impacts, profound changes in both production and consumption of food are needed, with the latter requiring a shift towards environmentally sustainable diets (IPCC, 2019a; Willett *et al.*, 2019). Such diets are often defined as those with lower environmental impacts (e.g. Burlingame & Dernini, 2012), identified by several studies as diets with a low to moderate amount of meat and animal-based products and a larger share of plant-based foods than in current diets in many high-income countries such as Sweden (Chai *et al.*, 2019; Springmann *et al.*, 2018; Clark & Tilman, 2017; Martin & Brandão, 2017; Hallström *et al.*, 2015). Although these studies provide valuable insights into how the environmental impacts of diets can be reduced, most do not show whether the diets are sustainable 'enough', *e.g.* to reach environmental targets such as those in the Paris Agreement on limiting climate change. For this, the environmental performance of the diets needs to be evaluated against an absolute threshold beyond which they can be considered unsustainable. Such thresholds have been suggested by the EAT-*Lancet* Commission for the emissions and resource use of six environmental variables related to food production (Willett *et al.*, 2019) enabling comprehensive evaluation of food-related environmental sustainability.

However, although the EAT-*Lancet* framework includes a number of environmental variables, these do not capture all environmental issues related to the food system. Further, the boundaries in the framework are defined and applied on a global level, so applying these on a national level might overlook important aspects of the conditions where the majority of the foods in the diet are produced. Hence, the variables may need to be complemented by additional indicators to cover missing aspects.

Moving towards diets with lower environmental impact and reduced consumption of animal-based foods has been found to be challenging (e.g. Hartmann & Siegrist, 2017). To drive changes in dietary patterns, policy instruments have been identified as necessary and a climate tax on food consumption has been suggested in several European countries, including Sweden (SSNC, 2015; Lööv *et al.*, 2013), the Netherlands (TAPPC, 2020a) and Germany (TAPPC, 2020b). The effects of a climate tax have been modelled in scientific studies, with findings indicating potential to reduce food-related greenhouse gas (GHG) emissions (e.g. Springmann *et al.*, 2016; Säll & Gren, 2015; Edjabou & Smed, 2013; Wirsenius *et al.*, 2011). However, several questions on the evaluation of a climate tax remain unanswered.

An important criterion when designing policy instruments is cost-efficiency, *i.e.* achieving emissions reductions at the lowest possible cost. In economic theory, a criterion for a cost-efficient tax is that all activities causing negative externalities, *e.g.* climate impact, should be taxed at the same level, irrespective of source and sector. For example, 1 kg of CO_2 from industrial activities should have the same tax rate as 1 kg of CO_2 from road fuel combustion. Mitigation options should then be implemented where emissions can be reduced at the lowest cost (Baumol & Oates, 1988). Moreover, a cost-efficient tax on GHG emissions should reflect the cost of the damage caused by a marginal increase in emissions (Pigou, 1920). Based on this, climate taxes should ideally be imposed on all products on the market and the tax rates should be set by determining the amount of emissions caused during production and the marginal damage cost of the emissions. A tax on food consumption rather than production is generally suggested, as such a tax has the advantage that domestic and imported products are equally affected. This reduces the risk of production moving to other countries, leading to emission leakage and offset of national reductions (Säll *et al.*, 2020; Van Doorslaer *et al.*, 2015).

For fossil fuels, calculating the carbon dioxide (CO_2) emissions caused by combustion of *e.g.* 1 kg of petrol or diesel (*i.e.* the emissions to be taxed) is straight-forward, as it is determined by the carbon content of the fuel. For each kg carbon combusted, 3.1 kg CO_2 will be formed (Eriksson & Ahlgren, 2013). For foods, however, there is no such inherent physical property of 1 kg food determining the emissions caused by that kg food. Instead, the emissions caused throughout the life cycle of the foods, *i.e.* from production, need to be determined for taxation.

For a cost-efficient tax, the emissions should ideally be measured at the point source. Food production involves a long chain of activities with GHG emissions of CO_2, methane (CH_4) and nitrous oxide (N_2O) arising in primary production on the farm, in production of farm inputs (*e.g.* fertilisers and feed), and in processing and distribution of foods. Additionally, the use of refrigerants cause emissions of fluorinated GHGs (Crippa *et al.*, 2021). Monitoring these emissions *e.g.* at farm-level would be costly and challenging, so food-related emissions are instead often modelled and weighted according to their impact on the climate using Life Cycle Assessment methodology (LCA).

In a policy context, LCAs should aim for robustness and consistency, and for providing affected stakeholders with transparent information (McManus *et al.*, 2015; Wardenaar *et al.*, 2012). Challenges to acquiring robust results when using LCA include *e.g.* choices in methodology and inventory data. For foods in particular, using LCA involves considering a number of methodological choices which can strongly affect the results. Further, food systems are inherently more variable than other sectors due to *e.g.* numerous agricultural holdings with differences in *e.g.* climate, soil conditions and management practices (Notarnicola *et al.*, 2017). As such, careful consideration should be taken when establishing climate impact data with LCA to use as a base for taxation. To achieve a cost-efficient tax, calculations should reflect as accurately as possible actual emissions from the life cycle of the foods. Apart from cost-efficiency, other important criteria in the design of policy instruments include simple administration (also affecting the costs) and acceptance by affected stakeholders.

Previous studies modelling the effects of a climate tax on food have primarily focused on possible reductions in climate impact. Changing dietary patterns to reduce meat consumption and increase consumption of plant-based foods has been found to have potential in reducing GHG emissions and land use (Hallström *et al.*, 2015), decreasing water consumption (Aleksandrowicz *et al.*, 2016) and alleviating pressure on eutrophication and biodiversity (Martin & Brandão, 2017). Thus a climate tax could potentially reduce several burdens simultaneously. However, reducing consumption of climate-burdening food such as beef has also been indicated to increase water consumption if substitution is made involving *e.g.* poultry (Martin & Danielsson, 2016) and to have higher impacts on freshwater ecotoxicity if *e.g.* pork consumption increases (Nordborg *et al.*, 2017). Further, in the context of Sweden, reduced beef consumption leading to fewer grazing animals has been highlighted as a potential hazard to biodiversity conservation of Swedish semi-natural pastures and threatened species within these (e.g. Lööv *et al.*, 2013). Therefore a climate tax could also lead to goal conflicts between reduced climate impact and other environmental aspects. More knowledge of how climate taxation affects environmental categories other than climate impact is therefore needed in order to identify potential trade-offs between affected environmental categories following introduction of a tax.

2 Aim and Structure

2.1 Aim and objectives

The overall aim of this thesis was to extend knowledge of how the environmental pressures of foods can be calculated for use in climate taxation and assessments of environmental sustainability of diets.

Specific objectives were to:

- Establish consistent and transparent datasets on the climate impact of foods for use in a climate tax
- Determine how methodological choices affect climate impact values and tax levels
- Evaluate how well global indicators capture local environmental sustainability concerns and assess the environmental sustainability of food consumption in a Swedish context
- Investigate the environmental effects of taxation and identify potential goal conflicts between environmental aspects as a result of taxation.

2.2 Structure and context of the work

2.2.1 Structure of the work

The thesis is based on four papers (I-IV) (Figure 1). The focus in Papers I-II was on the climate impact of foods. These two papers explored aspects related to establishing climate impact values for foods to be used in taxation, including how methodological choices affect results of climate impact values and tax levels. To this end, a method was developed in Paper II for calculating consistent and transparent datasets on the climate impact of foods, for use in a climate tax. The resulting datasets on the climate impact of foods were used in Papers III and IV, where the focus was extended to include environmental aspects other than climate impact and to assess the environmental sustainability of both foods and diets. Data on the environmental pressures for other environmental categories were also produced. In Paper III, aspects and indicators for assessing the environmental sustainability of food consumption in a Swedish context were identified, and the environmental sustainability of the current average Swedish diet was benchmarked against global boundaries. In Paper IV, the indicators identified in Paper III were used to evaluate the environmental effects of taxation and to identify potential goal conflicts from taxation.

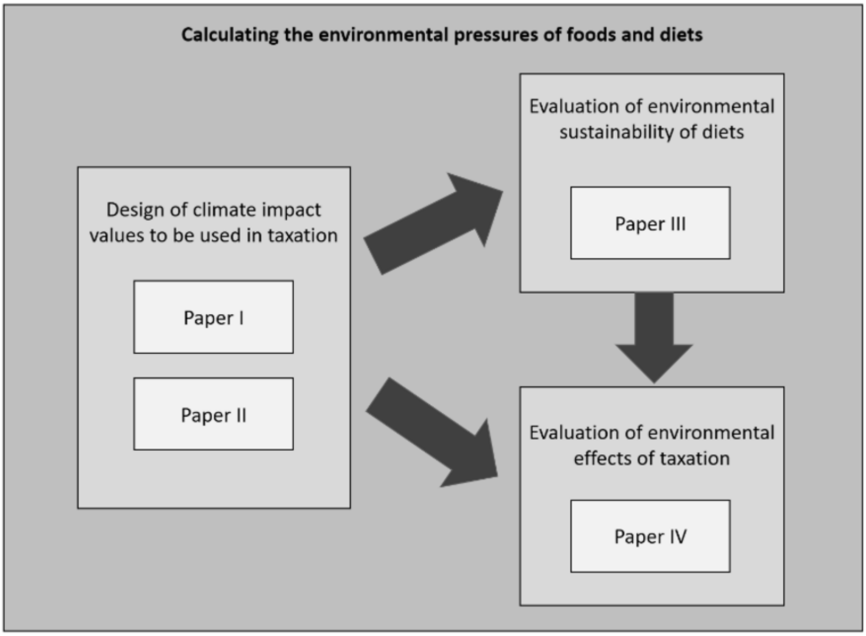


Figure 1. Structure of the work performed in Papers I-IV in this thesis.

2.2.2 Context of the work

The work described in this thesis was carried out within a transdisciplinary research project run in collaboration by the Department of Energy and Technology and the Department of Economics at the Swedish University of Agricultural Sciences. The project, which is described in Röös *et al.* (2021), had the overall objective of extending knowledge of how a climate tax on food consumption can be designed and to evaluate the effects of such a tax. The aim of the work performed at the Department of Energy and Technology is described in section 2.2.1 of this thesis.

In the work performed at the Department of Economics, the focus was on designing cost-efficient taxation, evaluating the effects of taxation on climate impact and analysing social and economic effects of taxation. The work included investigating the design of cost-efficient taxation, as described in Gren *et al.* (2019)[1]. A central component of the work was building a demand system to estimate how consumers change their consumption due to food price changes resulting from taxation, and evaluating the resulting effects from taxation on food-related climate impact (described in Säll *et al.* (2020). Moreover, by using the demand system, social and economic effects were estimated, including distributional effects on consumers (described in Säll (2021) and the nutrient intake of the population (described in Röös *et al.* (2021).

[1]As described in section 2.2.1, the work carried out within this thesis included establishing climate impact values and analysing the effects of methodological choices on resulting climate impact values and tax levels.

3 Background

3.1 Environmental impacts from the food system

3.1.1 Climate impact

Food-related activities account for about one-third of global anthropogenic GHG emissions. In comparison with the transport and energy sectors, where emissions of GHGs are dominated by fossil fuel-related CO_2, the emissions generated in the global food system also include substantial amounts of CH_4 and N_2O (Crippa *et al.*, 2021). Although large variations in the contribution to overall emissions have been reported, agriculture and associated land use and land use change (LULUC) activities have been found to dominate (Crippa *et al.*, 2021; IPCC, 2019a; Poore & Nemecek, 2018). For example, Crippa *et al.* (2021) estimated that one third of global food system emissions derive from biological emissions in agriculture, including CH_4 emissions from digestion in ruminant livestock, CH_4 and N_2O emissions from manure management, N_2O emissions from fertilised soils and CH_4 emissions from flooded rice cultivation (Figure 2). In the study by Crippa *et al.* (2021), a further one third of global food system emissions were estimated to arise as CO_2 due to temporary or permanent changes in land use and management. Similarly, IPCC (2019a) and Poore and Nemecek (2018) estimated that 75% and 80%, respectively, of emissions from the global food system are associated with agriculture and LULUC.

The remaining emissions arise from production of fertilisers, pesticides and capital goods used in agriculture, and in processing, refrigeration, packaging, storage, transportation, consumption and waste management of food commodities (Crippa *et al.*, 2021; IPCC, 2019a; Poore & Nemecek, 2018). Moreover, the use of refrigerants causes emissions of fluorinated GHGs, which are estimated to account for about 2% of global food system emissions (Crippa *et al.*, 2021). An example of a fluorinated GHG is the hydrochlorofluorocarbon R22 (HCFC-22), which is emitted through the use of refrigerants, *e.g.* for storing wild-caught fish on vessels (Ziegler *et al.*, 2013).

The findings reported by Crippa *et al.* (2021) show differences in energy intensities in the overall food system between developing and industrialised countries, with the latter associated with more energy-intense production. Energy use accounts for one third of overall food-related emissions in industrialised countries, compared with one fifth as a global average.

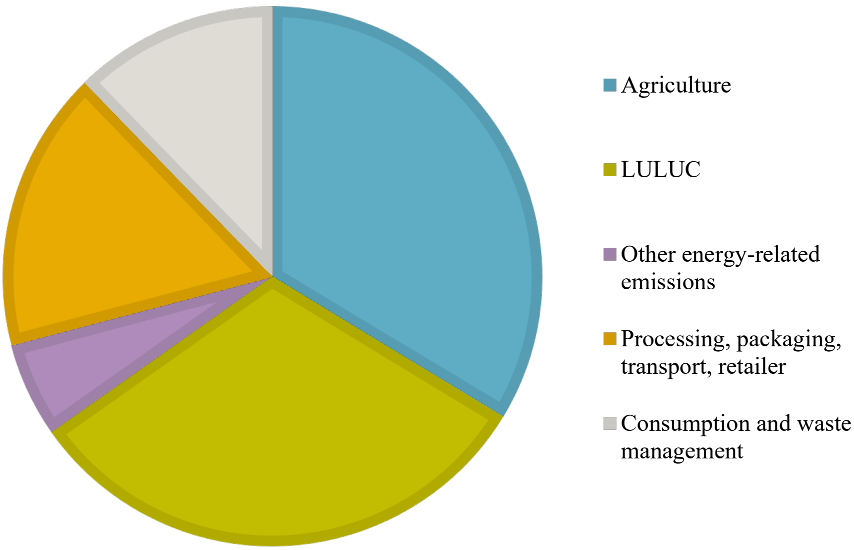


Figure 2. Emissions of greenhouse gases from the global food system in 2015. Source: based on Crippa *et al.* (2021).

3.1.2 Other environmental impacts

Apart from generating GHG emissions, food production also has other environmental impacts. The area used for crop cultivation and pasture for grazing animals occupies nearly 40% of the Earth's land surface (FAO, 2021) and expansion of agricultural land is the main driver of tropical deforestation (Pendrill *et al.*, 2019; Houghton, 2012). Expansion of land for food production has been identified as main driver of terrestrial biodiversity loss (Tilman *et al.*, 2017). As regards marine biodiversity, about 30% of global fish stocks are estimated to be overfished and 60% to be fully fished. Moreover, the agriculture sector is responsible for chemical pollution through the use of plant protection substances and accounts for 70% of global freshwater withdrawals, mainly for irrigation of crops (Willett *et al.*, 2019). Crop production and animal husbandry were previously closely connected by recirculation of nutrients from animal manure to growing crops. However, the use of mineral fertiliser has led to geographical separation of the production systems, which has caused disruptions in nutrient recycling (e.g. Billen *et al.*, 2013). As a consequence, increased use of nutrients in agriculture is causing increased losses of nutrients to the environment, leading to eutrophication of both terrestrial and aquatic ecosystems (Galloway *et al.*, 2003).

3.2 Options to reduce environmental impacts from the food system

A number of studies have identified a need for both production- and consumption-side strategies in order to reduce environmental impacts from the food system (e.g. IPCC, 2019a; Tilman & Clark, 2014; Foley *et al.*, 2011). Production-side measures include higher crop yields, improved management of land, fertiliser and manure, reduced use of fossil fuels and reduced deforestation (IPCC, 2019a). Consumption-side measures include *e.g.* reducing food waste and over-consumption of food, and changes in dietary patterns (IPCC, 2019a; Willett *et al.*, 2019). The focus in this thesis is on food consumption and consumption-side measures through dietary changes.

3.2.1 Policy approaches to reduce environmental impacts from the food system

A range of policy approaches can be applied to reduce environmental impacts from food consumption, including restrictive legislation, informative policies and economic incentives (Röös *et al.*, 2020; Garnett *et al.*, 2015). Informative instruments include awareness raising, labelling of environmental impacts and 'nudging', all of which aim to influence individuals in their choices (Röös *et al.*, 2020). For example, Swedish initiatives in the food sector include dietary guidelines with environmental considerations issued by the Swedish Food Agency (2019), 'climate certification' and labelling of the climate impact of foods (e.g. Estrella, n.d.; mat.se, 2021; Oatly, 2021; Svenskt Sigill, 2019) and the more comprehensive sustainability declaration by Coop (Coop, 2021). Informative policies can influence people's dietary choices (Edenbrandt *et al.*, 2021; Elofsson *et al.*, 2016), but have limited effectiveness in isolation. For example, a majority of consumers actively ignore climate information on foods (Edenbrandt *et al.*, 2021).

Economic policy instruments include taxes and subsidies to change the relative price of products and steer consumption in a desired direction. No tax has been levied specifically on food products for environmental reasons, but excise taxes on food to promote healthy eating exist, *e.g.* taxation on sugary drinks in several countries world-wide (Statskontoret, 2019). Further, food in the European Union (EU) is taxed within the value-added tax (VAT) system, but in several countries, including Sweden, the VAT rate on foods is lower than that on other goods and services. Hence, while the current standard VAT rate on goods and services in Sweden is 25%, products that are seen as particularly necessary for consumers have a VAT rate of either 12% or 6%. For example, books, cultural events and personal transportation have a VAT rate of 6% while all foods sold in Swedish retail outlets and restaurants have a rate of 12% (Swedish Tax Agency, n.d.-b). Changes to VAT rates to promote healthy eating have been implemented in countries such as the United Kingdom, where 'unhealthy' foods such as ice cream, confectionary and sugary drinks are targeted by a higher VAT rate than other foods (Röös *et al.*, 2020).

Although there is no specific environmental tax targeting food products, over one fifth of global GHG emissions are included in economic policy worldwide (World Bank, 2021) and in Sweden (Författningssamling, 1994). Carbon taxes or emission trading systems (ETS) cover CO_2 emissions from the use of fossil fuels, as well as N_2O emissions from mineral fertiliser production in the EU, which are included in the EU ETS. Thus, these policy instruments cover some of the emissions associated with food production. In Sweden, a deduction of 20% is made for the CO_2 tax for specific sectors, including agricultural processes and energy use for heating greenhouses (Swedish Tax Agency, n.d.-a).

3.2.2 Previous studies on climate taxation on food

The effects of climate taxation on food consumption have been modelled in multiple scientific studies, based on historical price and consumption changes (Broeks *et al.*, 2020; Forero-Cantor *et al.*, 2020; Zech & Schneider, 2019; Bonnet *et al.*, 2018; Chalmers *et al.*, 2016; Springmann *et al.*, 2016; Säll & Gren, 2015; Edjabou & Smed, 2013; Wirsenius *et al.*, 2011). Among these, Säll and Gren (2015) found that emissions from the Swedish livestock sector could be reduced by 12% with a Swedish consumption tax targeting animal products. Wirsenius *et al.* (2011) found that GHG emissions in EU agriculture could be

reduced by 7% with a tax on meat, dairy products, cereals and vegetables. Springmann *et al.* (2016) modelled global pricing of food based on the climate impact and identified potential to reduce global food-related GHGs by 9%.
In summary, the results of modelling studies vary depending on the country or region of implementation, the food products targeted by taxation and tax levels used in the simulations.

Apart from climate taxation, some studies have also investigated the potential for differentiating VAT rates in order to steer consumption in a desired direction, *e.g.* increasing the tax rate on foods with a high climate impact or lowering the VAT rate on foods with a low impact (Broeks *et al.*, 2020; Ekvall *et al.*, 2016).

3.2.3 Considerations of climate taxation and climate impact values to be used in taxation

A key factor when implementing environmental policy is cost-efficiency, which requires the policy of interest to target the environmental problem, *i.e.* GHG emissions, at the lowest possible cost (Government of Sweden, 2009). As mentioned above, a cost-efficient tax is obtained when all GHG emissions are priced at the same level as mitigation options can be implemented where emissions can be reduced at the lowest cost (Baumol & Oates, 1988). Further, the tax should correspond to the marginal damage cost of emissions, for which taxes ideally should reflect as closely as possible the emissions generated from production (Pigou, 1920). A cost-efficient food tax would then be determined by the amount of emissions caused by production of the food and the marginal damage cost of the emissions.

As further mentioned above, emissions caused by production of products and foods are often calculated using LCA methodology. For a cost-efficient tax, the calculations should ideally reflect accurately the varying emissions generated when producing different products using different technologies. This also includes considering existing taxes on CO_2 emissions from the use of fossil fuels, as well as the CO_2 and N_2O emissions included in the EU ETS system. If these emissions were targeted by additional taxation, it would be cheaper to decrease emissions elsewhere, so such a tax would not be cost-efficient. Thus, for a cost-efficient climate tax, only the GHG emissions that arise in the life cycle of foods but are not yet targeted by taxation should be included in the calculations.

Apart from cost-efficiency, another important criterion in the design of policy instruments is simplicity of administration (Government of Sweden, 2009). In the case of food, calculating the climate impact of the thousands of products on the global market produced using different technologies, and considering what parts of value chains are already taxed, could generate a heavy administration burden and high costs. Based on this, it could be necessary (at least in the short term) to base a tax on aggregated food groups which reflect an average of the climate impact caused to produce certain foods on the Swedish market, rather than on detailed LCAs for each and every food product on the market. Taxes could be differentiated based on different technologies used during production, but differentiating taxes on the country of origin of the product might not be a viable option, as that could violate trade agreements by the World Trade Organization (WTO). According to the 'most-favoured nation' principle, members of the WTO must apply the same trading rules to all other WTO members, so differentiated tax rates for different countries might risk being discriminatory (Bähr, 2015).

Further, when using LCA methodology for establishing climate impact values to be used in taxation, careful consideration has to be taken of methodological choices and inventory data. For robustness, credibility and simplicity of administration, the methodology should ideally be consistent across the foods to be taxed, and the results should be transparently presented.

Yet another relevant criterion in a policy context is for a tax to be accepted by affected stakeholders (Harring *et al.*, 2019; Drews & Van den Bergh, 2016), which for a climate on food consumption would primarily include consumers, producers and politicians. Drews and Van den Bergh (2016) discuss the level of policy support depending on a range of aspects, including socio-psychological factors of individuals (*e.g.* political orientation), type of policy instrument and contextual factors. According to Harring *et al.* (2019), public support for policy might be lower for taxes directed towards private consumption than for taxes directed towards the producer side. Drews and Van den Bergh (2016) suggest that individuals are more likely to accept policies which encourage rather than discourage behaviours, *e.g.* subsidies rather than taxes or regulations, and attribute this to the lower perceived behavioural and financial costs to the individual. Among contextual factors, Drews and Van den Bergh (2016) indicate increased support for policy in countries with a general trust in society, politicians and researchers. Harring *et al.* (2019) present evidence that a policy targeting fossil fuels is likely to be less well supported in countries with high economic dependency on the fossil fuel industry. Similarly, Harring (2020) reports that people living in rural areas in Sweden tend to be less positive to a meat tax than people living in big cities, which could be explained by the perceived risk of a meat tax to production of food in rural areas. Other important factors linked to acceptance include perceived policy fairness, where acceptance could increase if richer members of society pay a larger share and potential revenues are recycled to poorer or more vulnerable groups in society (Drews & Van den Bergh, 2016).

In summary, based on the above discussion, the climate impact values upon which a tax is based should ideally:

- Reflect the (ideally untaxed) climate impact of foods available on the market (*e.g.* in Sweden) to obtain a cost-efficient tax
- Be transparent, to be easy to administer and update
- Be established using consistent methodology across the taxed products, to make the tax robust and easy to administer
- Be established with consideration of acceptance by affected stakeholders.

3.3 Life Cycle Assessment

3.3.1 Life Cycle Assessment methodology

The environmental impacts of food products are commonly evaluated using LCA, a quantitative method for assessing the environmental impacts associated with a product or service over its lifetime. The LCA methodology has been standardised by the International Organization for Standardization (ISO, 2006b; ISO, 2006a), which provides guidelines on the four iterative phases of an LCA:

- *Definition of goal and scope*, which includes a description of the system, the purpose of the study and the methodological choices
- *Life cycle inventory*, where data are collected on emissions and resource use associated with the system under study
- *Life cycle impact assessment*, where the inventory data are classified and characterised according to the environmental impact
- *Interpretation of results*, which includes *e.g.* an evaluation of the main outcomes of the study and a discussion on the uncertainty and sensitivity of the results.

Apart from the general ISO standardisations on LCA methodology, several other standards have been developed. These include the International Reference Life Cycle Data system (ILCD) handbook by the European Commission (2010), which is based on the ISO 14040 and 14044 standards, but contains more detailed descriptions and requirements on how to perform LCA studies. Building on the ILCD handbook, the European Commission has launched an Product Environmental Footprint (PEF) standard, with the objective of establishing a common methodological approach for assessing the environmental impacts of products available on the European market and for possible use in future policies (European Commission, n.d.). Based on the PEF standard, product category rules (PCR) have been developed to give guidance on how to assess the environmental impact of specific products or product groups, including specific food items (European Commission, 2018). For foods in the PEF framework, PCR have been established for dairy products, pasta, beer and wine.

Product category rules have also been established under the Environmental Performance Declarations (EPD) system (The International EPD System, n.d.). The PCR in that system build on assessing the average environmental impact of products in a global context, and the resulting EPDs are mainly used in business to business communication. In the EPD system, PCR exist for a variety of food groups such as fruits and nuts, fish and seafood, poultry meat and dairy products.

3.3.2 Environmental Footprint Assessments

Similarly to LCA, *environmental footprint assessments* are based on life cycle thinking and are used to account for the environmental pressures from human activities, *e.g.* from the production of products or services. Footprints have been used to study different environmental concerns such as GHG emissions (ISO, 2018), freshwater use (Hoekstra *et al.*, 2011), land use (e.g. Kastner *et al.*, 2012), nitrogen and phosphorus use (Galloway *et al.*, 2014; Leach *et al.*, 2012; Wang *et al.*, 2011) and biodiversity (Lenzen *et al.*, 2012).

In general, environmental footprint assessment focuses on evaluating the pressures from resource use or emissions, while the LCA methodology assesses the potential impacts of such pressures (Vanham *et al.*, 2019). For some footprints, however, the assessments can include an impact phase. For example, the water footprint framework includes an optional impact phase (Hoekstra *et al.*, 2011). Further, the carbon footprint is often used synonymously with a product's climate impact as calculated with LCA.

Apart from the ISO standardisation of general LCA methodology (ISO, 2006b; ISO, 2006a), the specific ISO Carbon Footprint standard 14067 has been developed on how to perform a Carbon Footprint Assessment (ISO, 2018). The standard is consistent with the general ISO standardisation on LCA, but also includes specific recommendations on methodological choices related to calculations on the climate impact of a product.

Other standards on how to perform carbon footprint assessments include PAS 2050 on GHG emissions developed by the British Standards Institution

(BSI, 2011; BSI, 2008). The BSI has also developed a standardisation of carbon footprint specifically for horticultural products (BSI, 2012). Further, a common approach for assessing the carbon footprint of milk and dairy products has been developed by the International Dairy Federation (IDF, 2015). Figure 3 illustrates the relationships between existing standards on environmental and climate impacts of products in general and food products in particular.

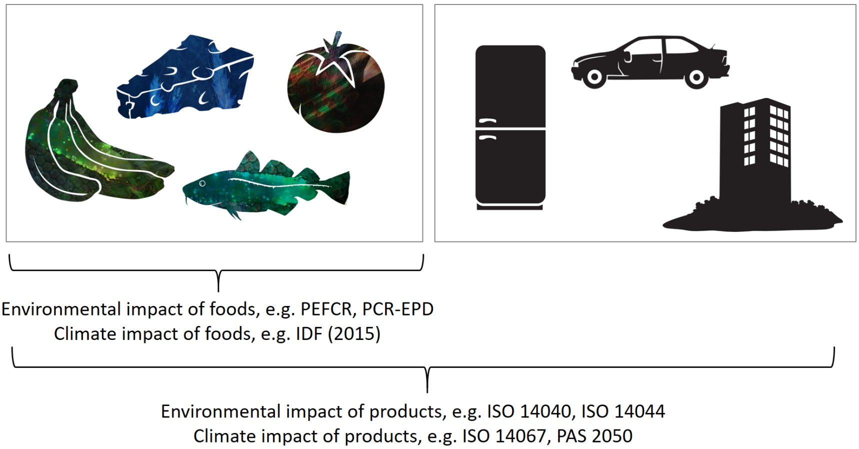


Figure 3. Relationships between existing standards on environmental and climate impact assessments of products in general and specifically of food products. Images used with permission from Fredrik Saarkoppel.

3.3.3 Methodological issues when using Life Cycle Assessment methodology to assess the environmental impacts of food

In the following, key concepts when using LCA methodology are described. Some of these are general concepts when assessing the environmental impacts of all types of products, whereas some are relevant specifically for food and agricultural products.

Functional unit

The functional unit describes the function of the studied product in a quantitative manner and is the reference to which the emissions from production are related. For food products, it is common to specify a functional unit with regard to production of one kg of a certain food product. However, use of a mass-based functional unit has been criticised by the LCA community for failing to consider the function of foods, which is mainly to provide an appropriate amount of different nutrients. Functional units based on the energy or nutritional content of the foods have therefore been proposed (Notarnicola *et al.*, 2017), and ‘nutrient indices’ which take into account the overall nutritional quality of foods have been developed (Hallström *et al.*, 2018). Such indices are commonly based on the content of a range of nutrients in food products in relation to the recommended daily intake of these nutrients. The various nutrient indices which have been developed differ with regard to the nutrients included (either to limit or encourage consumption), daily recommended values, whether to consider intake based on grams or kcal and the algorithm used to compute the index (Drewnowski, 2009).

System boundaries

The system boundaries define the processes that should be included in the study and vary according to the purpose of the assessment. In relation to food products, system boundaries can end at the farm (‘cradle to farm gate’) or at retail (‘cradle to retail gate’), or include all stages until consumption (‘cradle to mouth’ or ‘cradle to plate’) (Pernollet *et al.*, 2017). They can even include human digestion and waste management (Muñoz, 2021).

Accounting for emissions and sequestration due to soil carbon changes

Agricultural soils store a large amount of carbon in organic matter. Soils can be both sources and sinks of carbon, where the latter is beneficial from a climate perspective since CO_2 is removed from the atmosphere and stored in a less reactive form in soil (Freibauer *et al.*, 2004). Whether a soil loses or sequesters carbon is determined by various aspects such as management practices, biomass input, climate and soil characteristics. Soils acting as carbon sinks might have a high rate of sequestration in the early years after a change in management, *e.g.* introducing grass leys or catch crops into a cropping system dominated by monocropping of grains. However, the rate will decrease over time as the soil reaches a new equilibrium (IPCC, 2019a). Carbon storage in soils is also of a temporary nature, as factors such as changes in management practices may release carbon back into the atmosphere as CO_2 (Powlson *et al.*, 2011).

There are several ways of measuring soil carbon changes, but accurate determination of the changes requires long-term measurements (Röös & Nylinder, 2013). Instead, soil carbon changes are often estimated using models. The modelling approach is uncertain since the sequestration potential is highly variable between soil types, climates, management practices and, not least, the initial carbon content of soils and the reference soil used to compare against. Ideally, detailed site-specific data on field level are necessary to model changes, but the modelling is still highly uncertain (Keel *et al.*, 2017). Methodological developments are continually being made to account for soil carbon changes in LCA, but no scientific consensus has been reached (Bessou *et al.*, 2020). Due to the challenges, LCA studies on food and agricultural products may neglect to account for soil carbon changes (Goglio *et al.*, 2015). However, due to the strong influence of soil carbon changes on the results of climate impact assessments, accounting for these is recommended (e.g. Bessou *et al.*, 2020). Moreover, including soil carbon changes has been indicated to be of special importance in climate impact assessments of ruminants, due to the potential sequestration of carbon in pastures and in grass leys cultivated for feed, which can decrease the overall climate impact of the production system (e.g. Stanley *et al.*, 2018; Mogensen *et al.*, 2015; Pelletier *et al.*, 2010).

Among available models, the IPCC Tier 1 is a simplified approach based on fixed factors for different levels of land management, tillage intensity and inputs of organic material (IPCC, 2019b). The Introductory Carbon Balance Model (ICBM) (Andrén & Kätterer, 1997) accounts for emissions and sequestration of CO_2 in mineral soils based on variables such as climate and carbon input to soils from *e.g.* crop residues and manure. The ICBM is used in Swedish National Inventory Reporting.

Accounting for emissions of CO_2 from land use change

Land use change (LUC) refers to a change in the use of land by humans, which may lead to a change in land cover (Goglio *et al.*, 2015), *e.g.* when land is cleared through deforestation to be used for crop production or for grazing animals. Demand for agricultural land has been identified as a main driver of deforestation (Pendrill *et al.*, 2019; Houghton, 2012). Tropical deforestation in South America and Asia has been linked to exports of soybean and palm oil to Europe for use as animal feed, and is a highly debated issue in the European food sector (Karlsson *et al.*, 2021; WWF, 2014). Deforestation and other land use changes cause large GHG emissions and threaten biodiversity (Pendrill *et al.*, 2019; Tilman *et al.*, 2017).

Various methods for accounting for emissions from LUC are available but generate highly variable results, as the assessments rely on different assumptions about the drivers of LUC (Finkbeiner, 2014; Persson *et al.*, 2014). If land is cleared for arable production and crops are grown on the converted land, the conversion is commonly defined as direct LUC (dLUC) caused by the crops. Methods accounting for dLUC thus allocate the emissions from deforestation to the commodities being produced on the deforested land. As opposed to dLUC, indirect LUC (iLUC) describes how expansion in the production of a certain commodity displaces others in an area so their production is moved, which demands clearing of new and previously untilled land. Based on this approach, emissions from LUC are allocated to the expanding commodities in a certain area (Röös & Nylinder, 2013). Further, Audsley *et al.* (2010) suggest an approach based on the viewpoint that all demand for agricultural land contributes to LUC through its pressure on land use. Based on this, all global LUC emissions from agricultural expansion are allocated to total agricultural land area globally, resulting in one single LUC factor for all agricultural land. Hybrid LUC methods have also been established, *e.g.* Persson *et al.* (2014) suggest an approach where LUC emissions are estimated for commodities grown on recently deforested land, and based on the relative impact of the commodities on the overall expansion of agricultural land in a region. The method by Persson *et al.* (2014) calculates average LUC emissions from cultivation of agricultural commodities in different regions and countries.

Accounting for other biological emissions

Apart from CO_2 emissions from soils, as discussed in the sections above, biological emissions from agriculture also include CH_4 emissions from feed digestion in ruminant livestock, CH_4 and N_2O emissions from manure management and CH_4 and N_2O emissions from managed soils.

Emissions of CH_4 from enteric fermentation in ruminants are a by-product of feed digestion and are released to the atmosphere with the exhaled breath. Emissions of CH_4 also arise from manure stored in oxygen-free conditions and from anaerobically decomposed organic matter in flooded rice fields. Emissions of N_2O from soils and manure occur through the biological processes of nitrification and denitrification. In soils, these processes are accelerated by application of mineral fertiliser, manure and crop residues. The emissions are commonly divided into direct and indirect emissions, where the direct N_2O emissions arise in the studied agricultural soil, whereas the indirect emissions are caused by nitrogen lost from the soil by volatilisation, leaching and runoff (IPCC, 2019b).

Biological emissions are costly and challenging to quantify through measurements, due to large variations between emission sources depending on *e.g.* climate conditions and soil characteristics. Instead, these emissions are commonly estimated using models (Röös & Nylinder, 2013). Among available methods, N_2O emissions from managed soils can be estimated using the Tier 1 or Tier 2 methods developed by the IPCC, which assume that a fixed fraction of the nitrogen applied to mineral soils as mineral fertiliser, manure and crop residues is emitted as N_2O (IPCC, 2019b).

Environmental impact assessment method

To assess the environmental impacts of emissions and resource use in an LCA, an impact assessment method must be chosen for each environmental aspect. Depending on the environmental category, the available assessment methods may consider global or site-dependent impacts. For environmental categories such as climate change and stratospheric ozone depletion, impacts from emissions are global and therefore independent of emission site. For other impact categories, the emissions or resource use are often site-dependent and regionalising the impacts may therefore influence the results (Finnveden *et al.*, 2009). For example, the environmental impacts of water use can be assessed by accounting for the water availability globally or nationally. As water scarcity may vary substantially within a country, impacts of water use should ideally be assessed by accounting for the availability in a specific watershed (Boulay & Lenoir, 2020).

Another difference between impact assessment methods is whether they consider impacts on a 'midpoint' or 'endpoint' level (Finnveden *et al.*, 2009). For example, midpoint modelling of fertiliser use may assess the potential eutrophication of marine and terrestrial ecosystems, whereas endpoint modelling may evaluate the damage to biota or ecosystems caused by fertiliser use and subsequent eutrophication (Cosme & Hauschild, 2017).

With regard to climate impact, the impacts of different GHGs are commonly assessed by midpoint modelling in which the GHGs are weighted depending on their impact on radiative forcing (*i.e.* the net change in the energy balance of the Earth system), using the Global Warming Potential over a 100-year time horizon (GWP_{100}). The impacts of gases other than CO_2, such as CH_4 and N_2O, are weighted relative to the impact of CO_2, from which the weighted measure CO_2 equivalents (CO_2e) is obtained. In the 5th Assessment Report by the IPCC (Myhre *et al.*, 2013), GWP factors are provided both with and without the effects of climate-carbon feedback mechanisms, which include indirect effects on the carbon cycle from increased concentrations of GHGs that can further amplify (or dampen) climate change. One example is the reduced ability of soils and oceans to sequester carbon, which in turn increases atmospheric CO_2 and thus leads to amplified warming effects (Gasser *et al.*, 2017). In the latest (6th) Assessment Report by the IPCC, climate-carbon feedback mechanisms are routinely included in the GWP factors (Forster *et al.*, 2021).

The impacts of different GHGs can also be evaluated by assessing the GWP over different time horizons, *e.g.* 20 or 500 years. Further, the impacts from emissions can be evaluated using other metrics, such as the more economy-orientated Global Cost Potential (GCP) or Global Damage Potential (GDP) (Tanaka *et al.*, 2010), or by evaluating the impacts on temperature using Global Temperature Potential (GTP) (Shine *et al.*, 2005). When using the GTP metric, the impacts on temperature change are analysed for a certain time in the future. For example, GTP factors have been suggested to correspond to the time when the temperature target in the Paris Agreement (to limit warming to 2 °C) is expected to be met (Persson *et al.*, 2015).

Using a 100-year time horizon with *e.g.* the GWP metric in policy with a longer focus than 100 years has been pointed out as misleading, as it does not consider the climate impact from emissions after 100 years. This is important when there are contrasting impacts on temperature change, such as for short- and long-lived GHGs. An important short-lived GHG in agriculture is CH_4, which is broken down in the atmosphere after 12 years, in contrast to CO_2 which accumulates in the atmosphere. Thus constant emissions of CH_4 will cause no additional warming over time, as emissions and removals are approximately in equilibrium, whereas constant emissions of CO_2 will continuously add to climate warming since they accumulate in the atmosphere. Using the GWP measure over the fixed 100-year time horizon reflects the impact within the 100-year period, but these contrasting dynamics are not reflected after 100 years.

To better account for differences between GHGs, alternatives to GWP have been suggested, such as GWP* which gives weight to additional emissions of short-lived GHG (Lynch *et al.*, 2020). Using the GWP* metric in LCA of certain products is challenging, however, since it requires a decision on which emissions are additional, constant or decreasing.

3.4 Data availability on the climate impact of food

There is a large and emerging body of literature on the climate impact of food calculated with LCA methodology, ranging from assessments of specific food products to comprehensive reviews (e.g. Poore & Nemecek, 2018; Clune *et al.*, 2017). Further, a number of databases containing climate impact data for a variety of food products have been compiled, representing foods available on the global market. These include the World Food LCA database (available through the Ecoinvent database (Ecoinvent Centre, 2020), the thinkstep GaBi (thinkstep, n.d.) and the Agri-footprint database (van Paassen *et al.*, 2019). Databases have also been established covering the climate impact of foods on national markets, such as the Danish LCA Food Database (CONCITO, 2021), the French Agribalyse (Asselin-Balençon A., 2020), the Swedish RISE database (RISE, 2021) and the Swedish CarbonCloud (CarbonCloud, 2021).

Datasets and databases vary with regard to their transparency, where *e.g.* both the World Food LCA database and the Agri-footprint database allow the user to obtain data on GHG emissions, while *e.g.* the Swedish RISE database presents aggregated data on climate impact per kg food weighted using the GWP_{100} method. The Danish LCA Food Database and the CarbonCloud database present values weighted using the GWP_{100} method, but with transparency on the processes contributing to the overall impact. With regard to choice of system boundaries, these are commonly chosen to cover cradle to farm-gate emissions (*e.g.* Agribalyse, World Food LCA database, CarbonCloud). Some databases also add subsequent steps up to retailer level (*e.g.* RISE). Emissions or sequestration of CO_2 due to land use and/or LUC are accounted for in *e.g.* the World Food LCA, GaBi and Agri-footprint databases.

At the time of the thesis work, the only datasets representative of food available on the Swedish market were those offered by RISE (2021). These datasets have been compiled from various reports, conference proceedings and peer-reviewed studies, and therefore the methodology is not always consistent for the different products. As mentioned above, the datasets contain values of the climate impact per kg food, weighted using the GWP_{100} method, which limits transparency.

3.5 Frameworks for assessing environmental sustainability

Several frameworks have been developed to concretise or measure environmental sustainability. On a global level, the United Nations has established 17 Sustainability Development Goals and indicators to evaluate progress towards those goals (United Nations, 2015). Several goals are related to the food system and its associated environmental impacts. Moreover, the 'Planetary Boundary' concept developed by Rockström *et al.* (2009) and further refined by Steffen *et al.* (2015) defines thresholds for environmental impacts from all sectors on a global scale for nine Earth system processes. Building on the Planetary Boundaries concept, the EAT-*Lancet* Commission developed a framework for assessing the environmental sustainability of food systems (described in section 3.5.1). In Sweden, the Swedish Environmental Objectives framework aims at steering Sweden's environmental policy towards sustainable development (section 3.5.2).

3.5.1 The EAT-*Lancet* framework

The EAT-*Lancet* Commission, consisting of world-leading scientists from various disciplines including agriculture, environmental sustainability and human health, recently presented a comprehensive framework for assessing the environmental sustainability of the food system (Willett *et al.*, 2019). The Commission identified six Earth system processes specifically affected by food production. For each of these, a control variable was suggested and a global boundary within which humanity should operate to be environmentally sustainable was proposed (Table 1).

Table 1. *Description of the framework suggested by the EAT-Lancet Commission (Willett et al., 2019). E/MSY = extinctions per million species-years. Range of uncertainty for the global boundaries is given in brackets*

Earth system process	Control variable	Global boundary	Description of indicator
Climate change	GHG emissions	5 Gton CO_2e per year (4.7–5.4)	GHG emissions arising in food-producing activities.
Land-system change	Cropland use	13 million km^2 (11-15)	Cropland use for plant-based products and animal feed.
Nitrogen (N) cycling	N application	90 Tg N per year (65-130)	'New' reactive nitrogen from application of mineral fertiliser and nitrogen from biological fixation by plants.
Phosphorus (P) cycling	P application	8 Tg P per year (6-16)	Phosphorus from application of mineral fertiliser.
Freshwater use	Consumptive water use	2500 km^3 per year (1000-4000)	Groundwater and surface water used for crop irrigation and rearing of animals, which reduces flows in watersheds as it does not flow back to the same river or aquifer.
Biodiversity loss	Extinction rate	10 E/MSY (1-80)	Loss of potential endemic species of five taxa (mammals, birds, reptiles, amphibians, plants) from occupation of cropland and pastures.

3.5.2 The Swedish Environmental Objectives framework

The Swedish Environmental Objectives (SEO) framework was developed with the intention to steer Sweden's environmental policy towards solving environmental issues for the next generation, without causing environmental problems outside Sweden's borders (Sveriges miljömål, 2019). Deriving from this 'generation goal', 16 environmental quality objectives which reflect environmental concerns of importance for the Swedish context were established. For each objective, several indicators with different focal points are used (see Figure 4 for examples).

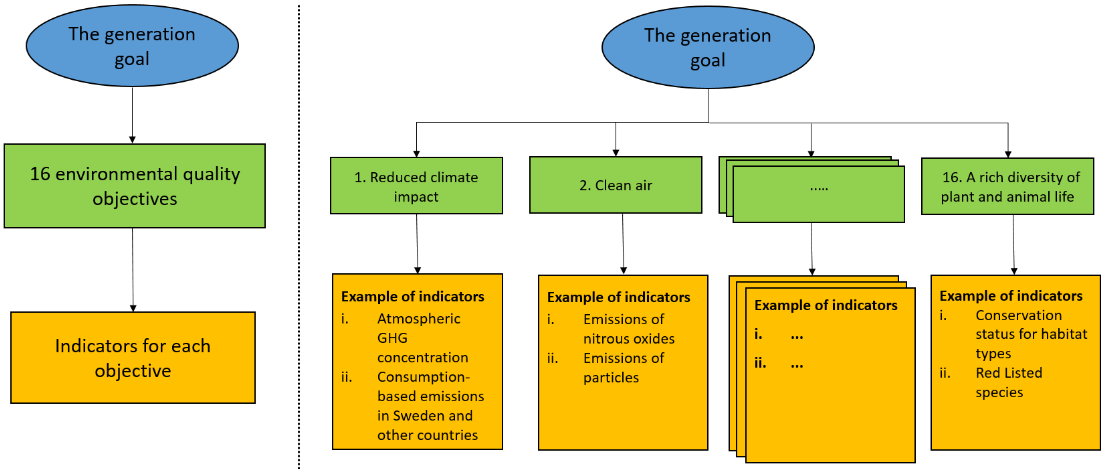


Figure 4. Illustration of the Swedish Environmental Objectives framework and examples of objectives and indicators. Source: based on Sveriges miljömål (2019) (diagram from Paper III).

4 Methods

This chapter provides an overview of the methods used in Papers I-IV. Section 4.1 focuses on establishing climate impact values for foods, to be used in taxation. None of the existing datasets on the climate impact of food, as presented in the scientific literature or in databases (described in section 3.4), was considered suitable to use in taxation or to study how methodological choices affect results of climate impact and tax levels. This was because none was simultaneously representative of the climate impact of foods available on the Swedish market, sufficiently transparent or consistent in the methodology across the food products to be used in taxation. Therefore, a method was developed in Paper II to establish consistent and transparent datasets on the climate impact of foods for use in a climate tax (the method is described in section 4.1.1). This method was applied in Papers I-II to test how methodological choices affect the results of climate impact values and tax levels (described in section 4.1.2).

Section 4.2 describes evaluation of the environmental sustainability of the Swedish diet in a local context. Section 4.3 focuses on methods for evaluating the environmental effects of taxation and potential goal conflicts. Finally, section 4.4 describes the food groups and data on food supply used for the calculations.

4.1 Establishing climate impact values of foods to be used in taxation

4.1.1 Developing a method for establishing the climate impact of foods for use as a base for taxation

The method developed in Paper II is based on LCA methodology (see section 3.3) and accounts for the average climate impact caused by production of foods available on the Swedish market. The climate impact was established for 52 food groups, such as 'beef', 'pork meat', 'cheese', 'potato' and 'flours'. These food groups were chosen to match the level of detail needed for evaluation of the effects of taxation (see section 4.4).

For each food group, the climate impact was calculated for the major production systems in countries representing 10% or more of the market in Sweden. A weighted average of the climate impact of each food group was then established using data on self-sufficiency produced by the Swedish Board of Agriculture (e.g. SBA, 2020; SBA, 2019) and import statistics by Statistics Sweden (2021) (see example for tomato in Figure 5).

To the greatest extent possible, the method accounts for the climate impact of foods using primary site-specific data, retrieved from official statistics in statistical databases (European Commission, 2021; FAO, 2021; SBA, 2021), National Inventory Reporting by the producing countries under the United Nations Framework Convention on Climate Change (UNFCCC, 2021) or national guidelines and reports from advisory services. When primary site-specific data were not available, data were collected in the following order of prioritisation based on availability from: the World Food LCA database; peer-reviewed LCA studies; and LCA reports. Standard values for emissions from transportation by sea and road were calculated through the NTMCalc Environmental Performance Calculator (NTM, n.d.-b). For other emission sources of less importance for the final results and for which primary data are usually not available, such as packaging and electricity use in processing, standard values representative for foods on the Swedish market were taken from peer-reviewed papers and unpublished reports. Waste factors were applied based on a report by the FAO (Gustavsson *et al.*, 2011).

The ISO 14067 carbon footprint standard[2] (ISO, 2018) was considered suitable to use as a basis for the modelling, as it permits use of consistent LCA methodology across all food products included in the calculations. Further, the standard is internationally recognised, which increases acceptability and trust.

[2]The ISO 14067 standard from 2013 was used as the basis for the calculations in Paper II. This standard has since been updated and this latest version (2018) is referred to in the thesis.

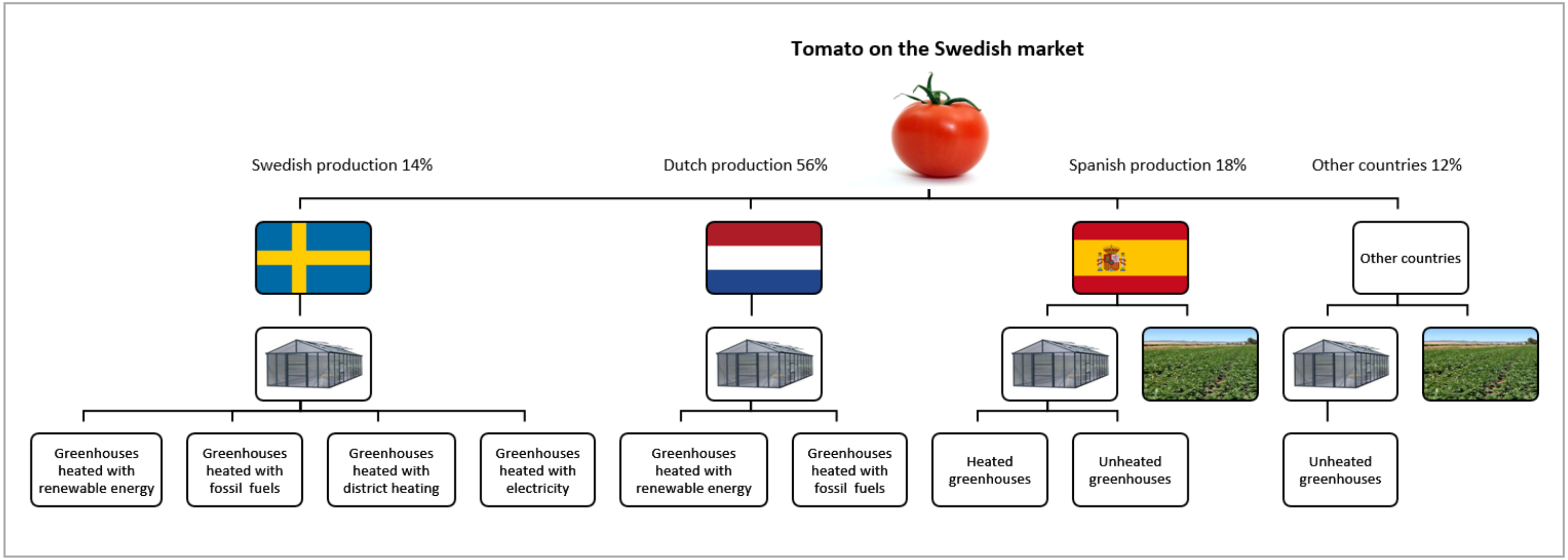


Figure 5. Assessment of the climate impact of tomatoes on the Swedish market using the method developed in Paper II, *i.e.* as a weighted average of the climate impact of production countries representing 10% or more of the Swedish market, with the major production techniques in the respective country (diagram from Paper II).

4.1.2 Methodological choices in climate impact values to be used in taxation

There are a number of methodological choices that need to be addressed in climate impact assessment based on LCA methodology, both in general and specifically for food products (section 3.3). Further, following the ISO 14067 carbon footprint standard (ISO, 2018), a number of methodological choices are recommended. However, as that standard was developed for climate impact assessments on products in general, and not on foods in particular, specific recommendations are not always given. In the context of taxation, methodological choices should also consider the criteria for a climate tax to be robust, cost-efficient, accepted by affected stakeholders and easy to administer (see section 3.2.3). Based on this, methodological choices were made according to the following.

Functional unit

The ISO standard does not specify any functional unit to be used in studies, but states that the functional unit should be consistent with the goal and scope of the study (ISO, 2018). If a climate tax targeting the whole market is set, the assessment of the climate impact needs to be related to the same functional unit across all foods. Thus using a functional unit based on one single function of foods (*e.g.* content of protein, fat or nutrients) would not be appropriate, as different foods provide very different functions. For example, while using per kg of protein as the functional unit could be suitable for protein-rich foods such as meat, dairy and legumes, it would not capture the benefits of fruit and vegetables, which are low in protein but dense in fibre and important micronutrients. Relating the climate impact of foods to a nutrient index would enable a broader assessment by including a range of nutrients. However, no estimates have been made of the damage costs of the nutrient content and environmental aspects combined, so cost-efficiency would be limited. Using per kg or litre as the functional unit would allow emissions to related to the estimated marginal damage cost of the emissions, which would be more in line with the theory of cost-efficient taxation.

In this thesis, the climate impact of foods was therefore calculated based on the functional unit of 1 kg of food (1 litre for drinks and oils).

System boundaries

Emissions that arise after retail gate in Sweden are covered by economic policy instruments, *e.g.* by the CO_2 tax on fuels (Författningssamling, 1994), the EU ETS system and the Waste Incineration Tax (Författningssamling, 2019). Ideally, emissions that are already taxed should be excluded when calculating values to be used in climate taxation. As such, these processes could be exempted from calculations when determining the basis for a climate tax (illustrated in Figure 6).

Within the retail gate perspective, emissions arising from the use of electricity and fuels include *e.g.* on-farm emissions from the use of machinery, heating of greenhouses and animal houses, and further in post-farm processes in the industry, as well as from transport. In Sweden and other European countries, many of these emissions are already targeted by taxation or included in the EU ETS system and these emissions should ideally be excluded in climate taxation.

Avoiding double taxation by accounting for current taxation is complex, however, especially as the origin and production methods of all ingredients in foods are commonly not known, requiring administration efforts. Including all emissions arising in production of foods from a 'cradle to farm gate' or 'cradle to retail gate' perspective is more administratively simple (Figure 6). An even simpler approach could be to limit the tax to emissions from biological processes (*i.e.* emissions from soils, enteric fermentation and manure management), which are currently untaxed and would thus not risk leading to double taxation (although this would lead to under-taxation, which is not cost-efficient).

In this thesis, emissions of the GHGs CO_2, CH_4 and N_2O and of the hydrochlorofluorocarbon R22 (HCFC-22) associated with production of the foods up to retail gate were accounted for (*i.e.* including emissions in the production of input materials, primary production, processing, packaging and transportation and including waste through all stages and at retailer). Emissions from minor emissions sources (*e.g.* production of pesticides and seeds, energy use for storage at wholesaler and retailer) were excluded from the calculations to simplify use of the method to establish climate impact values.

In Papers I-II, different system boundaries were used to investigate how the climate impact values changed and the implications for resulting tax levels (see Chapter 5). In Paper I, cost-efficient taxation was studied by adjusting for existing taxation, *i.e.* avoiding double taxation of emissions already targeted by taxation and applying the same marginal damage cost of emissions. The focus in Paper II was on analysing system boundaries to farm gate or retail gate, or to include only biological emissions. In all cases, the current Swedish CO_2 tax per kg CO_2e was applied. Emissions targeted by taxation with another tax rate were adjusted based on the difference between the Swedish tax and the current tax.

This included taxes in exporting countries and the 20% deduction on the CO_2 tax for the Swedish agriculture sector.

Accounting for emissions and sequestration due to soil carbon changes

In Sweden, emissions due to soil carbon changes are currently untaxed and sequestration of carbon in soils is not financially rewarded. According to the ISO 14067 standard, changes in soil carbon should be included in assessments of the climate impact of food based on an internationally recognised method or on a national approach described in a verified study (ISO, 2018). Due to the large uncertainties when accounting for soil carbon changes, it could be argued that such emissions should be excluded from climate taxation. However, as accounting for soil carbon changes can substantially affect the climate impact of food in general, and of ruminant meat in particular, including these emissions could be important for acceptance of the established climate impact of ruminant products.

In this thesis, soil carbon changes from land management were included using a simplified strategy based on the ICBM (Andrén & Kätterer, 1997). Based on the model, the 'soil carbon change potential' was calculated for cultivation of different crops, *i.e.* the amount of carbon a soil would hypothetically sequester or lose until reaching steady state compared with the current average carbon content in Swedish soils.

Accounting for emissions of carbon dioxide from land use change

There is currently no economic policy in Sweden targeting consumption-based emissions from LUC. The ISO 14067 standard states that emissions from dLUC should be accounted for in climate impact assessments. Based on the uncertainties of the drivers of iLUC, emissions from iLUC are recommended to be excluded in assessments until international consensus has been reached (ISO, 2018). However, none of the methods currently available for determining either dLUC or iLUC is generally accepted, and reaching consensus on methods will probably take a considerable time. Although the uncertainties in LUC calculations could favour exclusion of these emissions from a climate tax on food, it can be argued that they should be included as they are currently untaxed and that this could lead to higher acceptance of the established climate impact of foods associated with products known to drive deforestation.

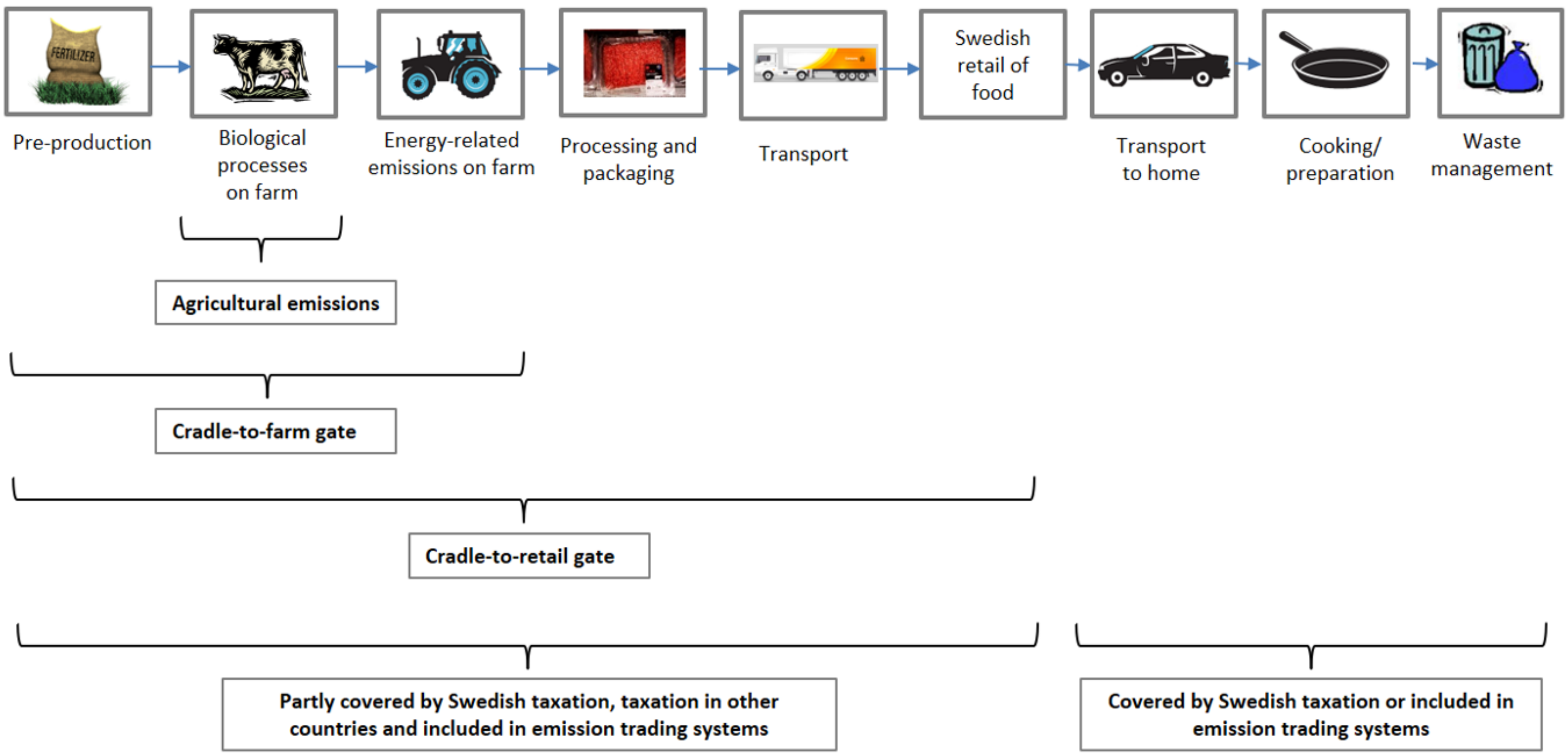


Figure 6. Illustration of different system boundaries to establish climate impact values for taxation (diagram modified from Paper II with images used with permission from Viktor Wrange and Fredrik Saarkoppel).

In this thesis, emissions from LUC were included for soybean and oil palm in animal feed, based on the approach suggested by Persson *et al.* (2014) and using emissions factors from Henders *et al.* (2015). This method was considered suitable as it represented the latest developments in accounting for emissions from LUC and as it calculates emissions on the average commodity from a certain region, not just the crops grown on recently deforested land.

Weighting emissions and tax levels of greenhouse gases

The ISO 14067 standard recommends using the latest GWP_{100} factors by the IPCC with climate-carbon feedback mechanisms included for summarising and weighting different GHG emissions (ISO, 2018). As GWP_{100} is the metric of choice in current climate policy, this would favour this method in calculation of data for use in a climate tax on food. However, to better account for the time when the temperature target in the Paris Agreement is expected to be met, other methods could be considered more suitable, such as the GTP factors suggested by Persson *et al.* (2015).

The associated cost of the damage caused by the climate impact should be applied to the climate impact values. The marginal damage cost of GHG emissions has been estimated for CO_2 alone (which is applied on weighted measures of CO_2e resulting from the GWP and GTP methods), but also based on the damage caused by individual GHGs (*e.g.* one price for emissions of CO_2, one for CH_4 and one for N_2O) (e.g. Marten *et al.*, 2015; Waldhoff *et al.*, 2014). Using a weighted measure such as GWP_{100} has been found to underestimate the damage cost of non-CO_2 GHGs, so using estimates of damage cost for individual GHGs has been argued to give more accurate results and lead to more cost-efficient policies (Marten *et al.*, 2015).

In this thesis, emissions were weighted using the GWP_{100} metric with factors from the latest IPCC report at the time of study, including climate-carbon feedback mechanisms (Myhre *et al.*, 2013). In Papers I-II, different metrics and marginal costs were used to investigate changes in the climate impact values and the implications for resulting tax levels (see Chapter 5). The alternative metrics included GTP_{100} and GTP with factors as suggested by Persson *et al.* (2015). To be consistent with current Swedish climate policy, the Swedish CO_2 tax was applied per kg CO_2e (equal to 1150 SEK/ton CO_2e (Swedish Tax Agency, n.d.-a; Swedish Tax Agency, n.d.-c). In addition, marginal costs for individual GHGs were applied based on different estimates from Marten *et al.* (2015) and Waldhoff *et al.* (2014). For consistency, the Swedish CO_2 tax was applied per kg CO_2, and the marginal damage costs of CH_4 and N_2O in Marten *et al.* (2015) and Waldhoff *et al.* (2014) were then adjusted in relation to the cost of CO_2. The marginal cost of CH_4 and N_2O in Marten *et al.* (2015) was 36 755 SEK and 450 515 SEK, respectively. In Waldhoff *et al.* (2014), the resulting cost per ton CH_4 and N_2O was 51 420 SEK and 708 721 SEK, respectively.

4.2 Assessing the environmental sustainability of the Swedish diet in a local context

In Paper III, aspects and indicators for assessing the environmental sustainability of food consumption in a Swedish context were identified (section 4.2.1). Based on this identification, the environmental pressures of foods were calculated in Papers III-IV (section 4.2.2) and the environmental sustainability of the Swedish diet was benchmarked against global boundaries in Paper III (section 4.2.3).

4.2.1 Identifying relevant environmental aspects and indicators for evaluating the environmental sustainability in a local context

In Paper III, the national indicators in the SEO framework were compared against the global indicators in the EAT-*Lancet* framework. Based on this, aspects and additional indicators were suggested to complement the existing indicators in the EAT-*Lancet* framework and capture local aspects of environmental sustainability of the Swedish diet. The analysis also revealed areas where additional data or method developments are needed, based on which indicators to be used in the evaluation of taxation in Paper IV were suggested.

4.2.2 Calculating the environmental pressures associated with Swedish food consumption

To calculate the environmental pressures of the Swedish diet, the pressures per kg or litre of food were assessed and then multiplied by the amount of food in the diet (for food supply data, see section 4.4). For each food group, the weighted average of environmental pressures associated with production of food available on the Swedish market was calculated, similarly as for the climate impact (see section 4.1). Table 2 lists the environmental aspects, indicators and inventory data used in calculation of the environmental pressures of foods in the Swedish diet.

Table 2. *Description of environmental aspects, indicators and inventory data used for calculating environmental pressures associated with Swedish food consumption (Papers III-IV)*

Environmental aspect	Environmental indicator	Description of inventory data
Climate change	GHG emissions (Papers III-IV)	See description of inventory data in section 4.1. For GHG emissions caused by production of foods not included in Paper II, land-specific data were collected in the following order of prioritisation based on availability: national official statistics; national guidelines and reports from advisory services; the World Food LCA database; peer-reviewed LCA studies; LCA reports but adjusted to match the methodology in Paper II.
Land-system change	Cropland use (Papers III-IV)	Calculated through yield levels for plant-based products and feed. Yield levels primarily obtained from national statistics (e.g. European Commission, 2021; SBA, 2021), official statistics from the Eurostat database (European Commission, 2021) or the FAOSTAT database (FAO, 2021), and otherwise retrieved from LCA studies or LCA reports.
	Pasture use (Paper IV)	Calculated through time on pastures and pasture use per animal. Data on time on pastures primarily obtained from the National Inventory Reporting by country (UNFCCC, 2021). Pasture use per animal obtained through advisory services, LCA reports and expert communication.
Nitrogen (N) and phosphorus (P) cycling	N and P application (Papers III-IV)	Data on fertiliser application rates collected in the following order of prioritisation based on availability: national official statistics (*e.g.* Statistics Sweden, 2021); national guidelines and reports from advisory services; the World Food LCA; peer-reviewed LCA studies; LCA reports. Data on biological nitrogen fixation rates by plants and pastures were obtained from the literature (Lassaletta *et al.*, 2014; Cederberg & Nilsson, 2004).
Freshwater use	Consumptive water use (Papers III-IV)	Data mainly obtained from the WaterStat database (Mekonnen & Hoekstra, 2012; Mekonnen & Hoekstra, 2011).
Biodiversity loss	Terrestrial extinction rate (Papers III-IV)	Characterisation factors were obtained from Chaudhary and Brooks (2018) on the loss of the different taxa, differentiated by country, for occupation of cropland and pasture. Data collection on cropland and pasture use described in the respective category.

Air pollution	Nitrogen oxides (NO_x) emissions (Paper IV)	Emissions of NOx from transportation by sea and road were calculated through the NTMCalc Environmental Performance Calculator (NTM, n.d.-a). International transportation were assumed to start at the capital of the production country and end in Stockholm, the capital of Sweden. Transport distances within Sweden were estimated through the simplified calculation model by the Swedish Climate Certification for Food (2010), and then calculated through the NTMCalc Environmental Performance Calculator (NTM, n.d.-a).
Acidification of freshwater and land	Ammonia (NH_3) emissions (Paper IV)	Emissions of NH_3 arising from application of mineral and organic fertiliser to fields, direct storage of manure, and losses via ventilation in animal houses. Data on fertiliser application rates described in the category of nitrogen and phosphorus application. Data on direct deposition of urine and manure on pasture and emission factors for the resulting NH_3 emissions retrieved from National Inventory Reporting by country or from official IPCC guidelines (IPCC, 2006).
Chemical pollution	Pesticide use (Paper IV)	Based on the amount of active ingredient in the pesticides. Data obtained from different sources in the following order of prioritisation, based on availability: country-specific statistics; country-specific data through guidelines or advisory services; country-specific data from the European Union of the average use of different crops or crop categories in the member countries (Eurostat, 2007); country-specific or average data from the World Food LCA database.
Ozone depletion	N_2O emissions (Paper IV)	See description of data on GHG emissions in section 4.1.

4.2.3 Benchmarking the environmental sustainability of the Swedish diet

In Paper III, the environmental sustainability of the Swedish diet was assessed by benchmarking the environmental pressures of the Swedish diet against the boundaries suggested by the EAT-*Lancet* Commission (for climate and land-system change, nutrient cycling, freshwater use and terrestrial biodiversity loss). The boundaries were defined on a global level and downscaled to per-capita boundaries for the global population, which offered insights into how much each global citizen uses of the globally ‘allowed’ emissions and resource use from the food system, regardless of where the pressures are caused.

4.3 Evaluating the environmental effects and identifying potential goal conflicts of taxation

In Paper IV, the environmental effects of taxation were evaluated and potential goal conflicts resulting from taxation were identified. The effects were modelled for a scenario where all food products on the Swedish market were targeted by taxation (section 4.3.1) and for a set of alternative taxation scenarios (section 4.3.2).

4.3.1 Taxation of all foods on the market

To evaluate the environmental effects of taxation, data were needed on consumer responses to price changes in food due to taxation (*i.e.* based on the climate impact values and using a tax rate of 1.15 SEK per kg CO_2e, corresponding to the 2015 Swedish tax on CO_2 emissions (Swedish Tax Agency, n.d.-a). The consumer responses were estimated using a demand system described in Säll *et al.* (2020) that estimates price elasticities for food groups in the Swedish diet by using historical price and consumption data on the foods. The elasticities show how consumers react to price changes, allowing for estimation of the effects of a tax.

Based on the simulated effects in each environmental category, potential goal conflicts resulting from taxation were identified.

4.3.2 Alternative taxation scenarios

How taxes are set is the result of political negotiations and compromises, and the final result will not always reflect what is 'optimal' or 'correct'. Thus it is possible that other climate impact values could be used in taxation and other tax alternatives than including all foods on the market could be applied. In Paper IV, different alternative taxation scenarios were modelled to evaluate the environmental effects of taxation and identify potential goal conflicts from taxation.

While a climate tax in theory would be cost-efficient if targeting all foods on the market, limiting a climate tax to include only the most high-impacting products, such as beef and dairy products, could lead to a decreased administration burden. From another perspective, beef cattle and other ruminants can contribute positively to food systems by maintaining biodiversity in pastures by grazing. Further, ruminants convert grass and other roughage which is inedible to humans into foods, while production of pork, chicken and eggs requires feedstuffs which could be used for human consumption (Van Zanten *et* *al.*, 2018; Röös *et al.*, 2016). For these reasons, it could be considered important to sustain ruminant meat production, while limiting consumption of monogastric meat and eggs through taxation.

Based on this, the scenarios modelled included different sets of food products in taxation, *i.e.* only animal products, only beef or only monogastric meat and eggs. For these scenarios, the effects of taxation were estimated based on the climate impact values combined with a tax rate of 1.15 SEK per kg CO_2e, corresponding to the 2015 Swedish tax on CO_2 emissions.

As an alternative to implementing taxation based on the climate impact of food products, changes to the existing VAT system could be used to steer consumption in a desired direction, *e.g.* increasing the tax rate on foods with a high climate impact or decreasing the rate for foods with a low impact (Broeks *et al.*, 2020; Ekvall *et al.*, 2016). Therefore, in Paper IV the environmental effects were modelled for scenarios involving changes to the VAT system (increasing the rate on animal products to 25%, reducing the rate on fruit, vegetables and cereals to 6%, or using a combination of both).

4.4 Food groups and data on food supply used for the calculations

In Paper II, the climate impact was established for food groups to match the level of detail in the demand system described in Säll *et al.* (2020), which was used for evaluation of the effects of taxation in Paper IV. As described in section 4.1.1, this included 52 food groups such as ‘beef’, ‘pork meat’, ‘cheese’, ‘potato’ and ‘flours’. In Paper III, the climate impact of a number of additional foods was calculated, to assess the impacts of the whole diet. This included a total of 98 food groups on the level of detail of *e.g.* ‘beef’, ‘pork meat’, ‘cured meats’, ‘canned meats’, ‘hard cheese’, ‘processed cheese’, ‘wheat flour’, ‘rye flour’ and ‘oats’.

Data on food supply for calculation of the environmental pressures of the diet in Paper III were mainly data obtained from the Swedish Board of Agriculture (SBA, 2021) on direct consumption of food in Sweden 2011-2015, *i.e.* the average amount of food available for consumption. For processed and prepared foods, lists of raw materials were primarily obtained from the Swedish National Food Agency (n.d.) or otherwise from LCA studies or reports. Data on food supply were considered suitable for use as they show the amount of food that needs to be produced to sustain the average Swedish diet. However, the data are not equivalent to actual consumption, as the food available for consumption could be eaten or wasted. Using results on food consumption from *e.g.* national dietary surveys (*e.g.* Amcoff, 2012) might give a better representation of the actual diet, but could underestimate actual intake due to underreporting.

For the calculations in Paper IV, data on food supply were a mixture of direct consumption and total consumption data from the Swedish Board of Agriculture (SBA, 2021), together with data (FAO, 2021). Data were retrieved from Säll *et al.* (2020), who concluded that a mix of consumption data was necessary in order to combine consumption levels with price indices to build their demand system.

5 Results and Discussion

5.1 Climate impact values and resulting tax levels

Using the method developed in Paper II, datasets representative of the climate impact of foods available on the Swedish market were produced using consistent methodology across the food products. In Papers I-II, different methodological choices affecting the climate impact values and tax levels were analysed. In this section, selected results building on the work in Papers I-II are summarised and discussed.

5.1.1 Climate impact values of foods on the Swedish market

Figure 7 shows the climate impact values established for a selection of food groups using the method developed in Paper II, and also variations in the climate impact values. The highest climate impact per kg was found for animal products (Figure 7a), especially beef, cheese and butter, while a substantially lower climate impact was found for plant-based foods such as fruits and vegetables, with the exception of products such as rice and coffee (Figure 7b). The variations in climate impact values represent differences in production countries and production systems, as further discussed in section 5.1.2.

The values are representative for the Swedish market, but the general finding that animal-based foods have a higher climate impact than most plant-based foods is also in line with findings in review studies on global LCA datasets, such as those by Clune *et al.* (2017) and Poore and Nemecek (2018). The results were also in line with previous findings for Swedish food products (e.g. Flysjö *et al.*, 2014; Davis, 2011; Cederberg, 2009; Berlin, 2002), when accounting for methodological differences such as adjusting waste factors, excluding emissions and/or sequestration from soil carbon changes and LUC, and weighting of GHGs using GWP factors from the 4th IPCC report (Forster *et al.*, 2007).

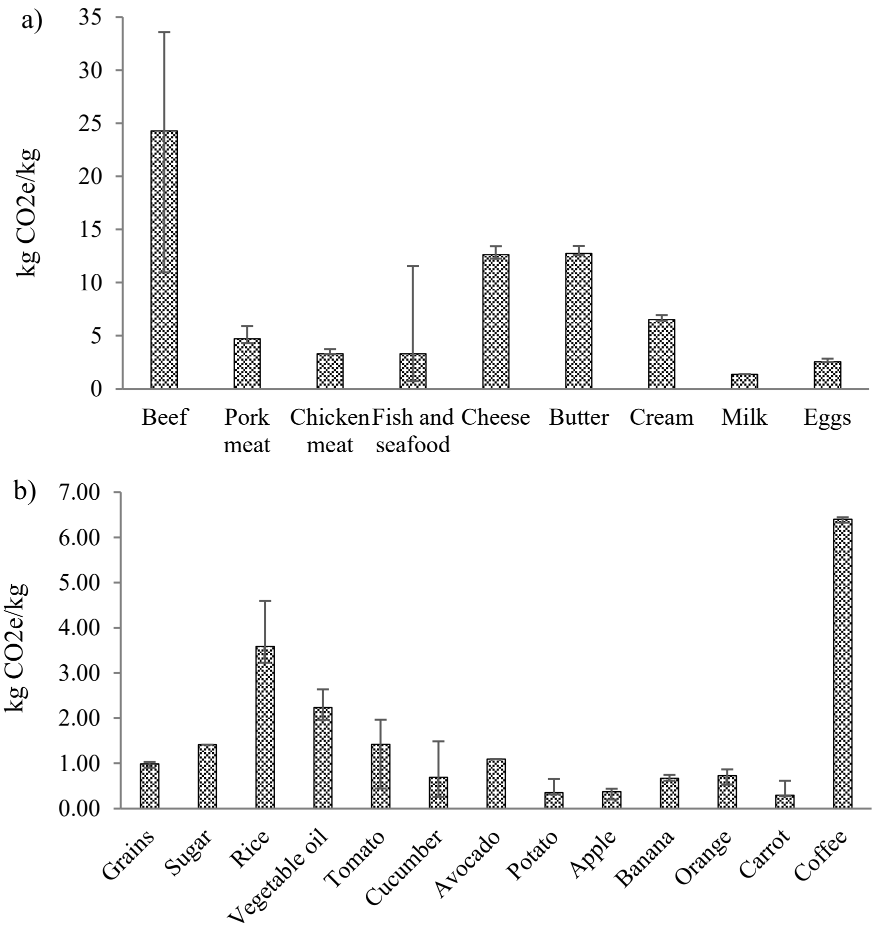


Figure 7. Climate impact values and variations* in impacts calculated with the method developed in Paper II, representing the impact per kg or litre of foods available on the Swedish market for a) animal-based products and b) plant-based products. Meat and fish are presented as carcass weight/with bones. *The variations in climate impact values represent differences between countries and production systems (see section 5.1.2).

5.1.2 Variations in climate impact values and level of detail of tax levels

Large variations were found for the climate impact within product groups such as fish and seafood, beef, and vegetables cultivated in greenhouses and open field (Figure 7). The variations in the climate impact for fish and seafood were explained by the large diversity of species within that food category. The highest impact was identified for wild-caught shrimp (northern prawn) and flounder (European plaice), while *e.g.* herring and mackerel were identified as having a considerably lower impact. These general findings have also been made in previous studies, such as that by Gephart *et al.* (2021).

For beef, the variation was mainly explained by differences in production systems. In line with previous studies (e.g. De Vries *et al.*, 2015), beef produced in dairy systems (*e.g.* from culled dairy cows and their offspring raised for meat) was found to have a substantially lower climate impact than beef produced in suckler-based systems, as the climate impact in the dairy system is allocated between the beef and milk that are jointly produced in more intensive systems.

For vegetables produced in greenhouses (tomato and cucumber), the highest impact was found for production in heated greenhouses, where a large amount of fossil fuels is used in production.

In this work, food groups (*e.g.* fish and seafood, beef, tomatoes, cucumber) were chosen to match the level of detail in the demand system, for evaluation of the effects of taxation (see section 4.4). With regard to how a climate tax could be designed in practice, the level of detail of the food groups could be chosen differently. For example, taxes could be differentiated based on fish species or on different production systems, *e.g.* for beef in suckler-based and dairy systems or for vegetables cultivated in open fields, in greenhouses using mainly bio-based fuels and in greenhouses using mainly fossil fuels. In theory, taxes could also be differentiated on an even more detailed level, *e.g.* at individual food product level. As discussed in section 3.2.3, differentiating tax rates on a detailed level could potentially generate a costly administration burden. However, such a tax would be more cost-efficient, as the climate impact values would be more accurately represented. This could also create incentives for producers to improve their production.

For implementation in the near future, a tax could be based on average climate impact values, such as those presented in this thesis. If the administration system were to allow use of more detailed values, the tax system could be further developed to include this. For example, producers could document their climate impact by making a climate impact declaration (*e.g.* through an authorised third party, following standardisation), which could be used instead of the average value. This would be similar to existing reporting on the environmental performance of producers in other sectors, *e.g.* within the Renewable Energy Directive (RED) of the EU (European Union, 2009).

5.1.3 Changes in price due to taxation

The absolute and relative price changes due to taxation for a set of food groups are shown in Figure 8a and 8b, respectively. These changes are based on the climate impact values shown in section 5.1.1 and using a tax rate of 1.15 SEK per kg CO_2e, corresponding to the 2015 Swedish tax on CO_2 emissions.

The highest absolute and relative price changes were seen for beef and dairy products, due to their high climate impact. In comparison, price changes for plant-based products such as fruits and vegetables were found to be lower (Figure 8). However, plant-based products such as coffee, rice and grains demonstrated a higher price difference. For coffee and rice, this was explained by their higher climate impact per kg than other plant-based products. For grains, the high relative price change was mainly explained by their low initial price.

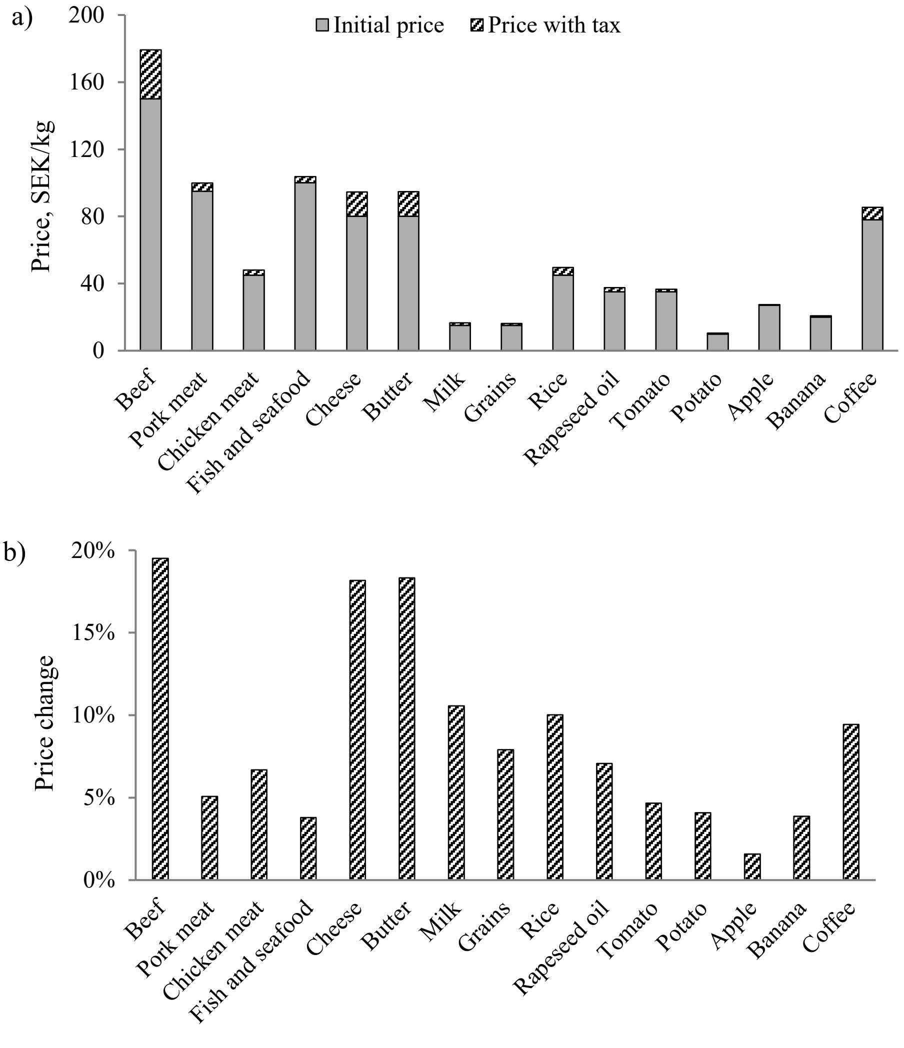


Figure 8. a) Price of different food products before and after taxation using a tax rate of 1.15 SEK per kg CO_2e, corresponding to the 2015 Swedish tax on CO_2 emissions, and b) percentage price changes due to taxation.

5.1.4 Influence on tax levels of choice of system boundary and adjustments for existing taxation

In Figure 9, the results of climate impact values depending on the choice of system boundaries (including emissions to farm gate, retail gate or only biological emissions) are shown for two food products from different production countries: beef from Swedish and Irish suckler-based production, and tomatoes from Swedish and Dutch production in heated greenhouses and from Spanish production in unheated greenhouses. These countries were the largest producers for the respective product on the Swedish market in the study period (2011-2015).

The results in Papers I-II revealed that the majority of emissions from beef production in both Sweden and Ireland are biological, including CH_4 emissions from feed digestion and CH_4 and N_2O emissions from manure management. Emissions from processing, packaging and transportation were found to account for a minor share (Figure 9a). For tomatoes, on the other hand, the results in Papers I-II indicated that biological emissions account only for a few percent in all three production countries studied (Figure 9b). The majority of the emissions from tomatoes produced in Sweden and the Netherlands were found to consist of CO_2 from energy use in greenhouses. Hence, the farm-gate modelling for those countries was similar to including all emissions to retail gate (corresponding to 90% of overall emissions). For Spain, the results revealed that the largest emissions source to be post-farm gate, *i.e.* transportation to Sweden, while emissions from greenhouse production are minor as production takes place in unheated greenhouses.

In Paper II, the relative emissions identified for beef production (*i.e.* a majority of emissions are biological, while post farm-gate emissions are minor) were found to resemble those for animal-based products such as milk and dairy products. For other meat types such as pork and chicken, emissions from energy use and fertiliser use for feed production were also found to be important. For plant-based products which are produced in open fields, a large part of the emissions was found to arise post-farm gate, *i.e.* in processing, packaging and transportation. An important exception was rice, where agricultural emissions of CH_4 account for a major part of the climate impact, due to high emissions from flooded rice paddy fields.

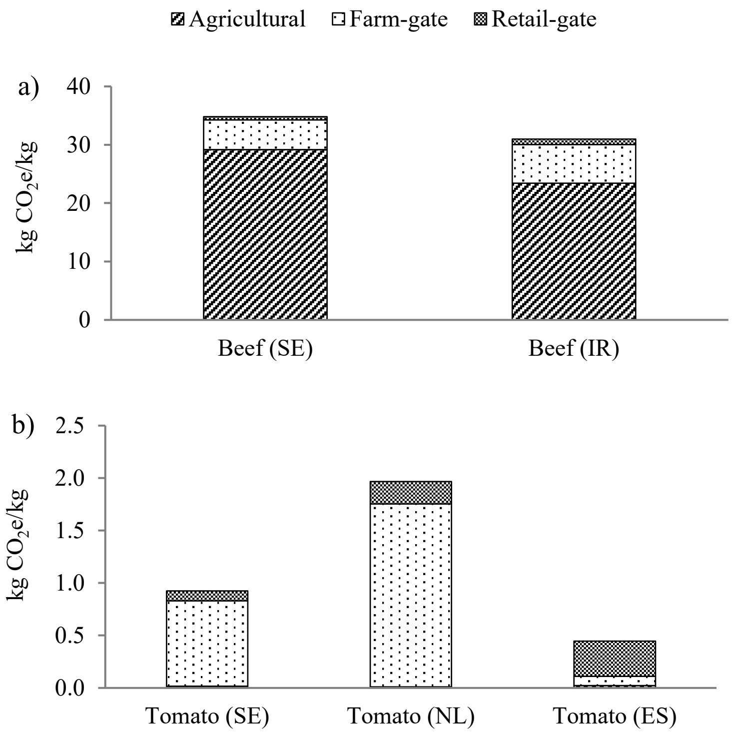


Figure 9. Climate impact values obtained with different system boundaries for a) beef and b) tomatoes. SE = Sweden, IR = Ireland, NL = the Netherlands, ES = Spain.

In Paper I, an analysis was made of the implications of calculating a theoretically cost-efficient tax by exempting emissions already targeted by taxation and adjusting all emissions to be targeted by the marginal cost of the Swedish tax on CO_2 emissions. Figure 10 shows the deviation between this 'optimal' tax and a tax including emissions using the different system boundaries (emissions to farm gate, retail gate or only biological emissions). This is illustrated as the ratio between the tax in the different scenarios and the optimal tax.

For beef, most of the emissions from production in both Sweden and Ireland currently remain untaxed, as economic policy instruments globally mainly focus on CO_2 emissions from the use of electricity and fuels, and not CH_4 emissions (section 3.2.3). Thus, for beef, the resulting tax levels for the different scenarios were all found to be close to the optimal tax, *i.e.* when correcting for taxed inputs and adjusting to the marginal cost of the Swedish tax on CO_2 emissions (Figure 10a). For tomatoes, a large proportion of the CO_2 emissions from heating greenhouses is subject to taxation in Sweden and in the

Netherlands. Furthermore, emissions from transportation are subject to taxation in Sweden and in other European countries. Thus the findings in Paper I indicated that if taxation were to include all emissions up to retail, the resulting tax on Swedish tomatoes would be more than three times higher than if correction were made for taxed inputs and marginal costs (Figure 10b). For tomatoes from the Netherlands and Spain, the difference to the reference tax was found to be less prominent. A large proportion of the emissions is already targeted by taxation, but the tax rates are lower in the Netherlands than in Sweden, so correction of the marginal costs is needed to obtain cost-efficient taxes.

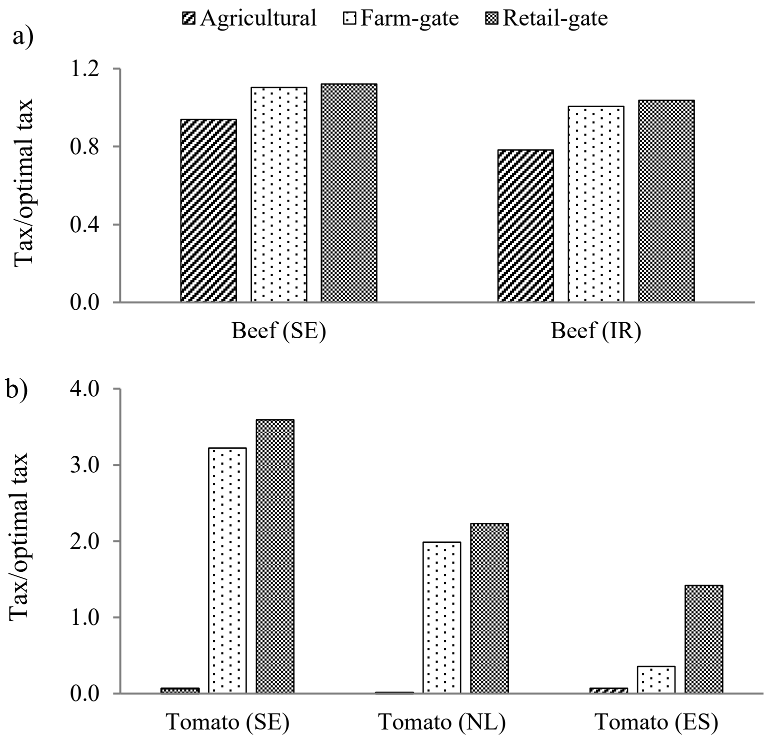


Figure 10. Ratio between tax levels with different system boundaries and the 'optimal' tax where adjustments for taxed inputs and marginal costs are made for a) beef and b) tomatoes. SE = Sweden, IR = Ireland, NL = the Netherlands, ES = Spain.

Although correction for taxed inputs and adjustments to the marginal cost would theoretically achieve a cost-efficient tax, this might require differentiated taxes on food products from different countries which, as discussed in section 3.2.3, might risk violating WTO trade agreements. Further, making these adjustment would probably lead to higher administration load and costs, as the Swedish market contains thousands of food products produced in various production countries (see discussion in section 5.1.2).

As an alternative, a tax could be implemented to target only biological GHG emissions. However, this would pose a risk of fossil emissions of CO_2 remaining untaxed in the case of imports from countries where CO_2 taxes are not in place, so choosing to implement a tax to either farm gate or retail gate might be more feasible.

In summary, choosing emissions to include in climate impact values used to set a tax involves a trade-off between ease of administration and accuracy of emissions levels.

5.1.5 Influence on approaches of weighting emissions and tax levels of different greenhouse gases

Figure 11 shows tax levels for a number of foods obtained using different metrics to weigh GHGs, *i.e.* using the GWP_{100} and the GTP_{100} metrics and including climate-carbon feedback mechanisms (Myhre *et al.*, 2013), as well as GTP factors suggested by Persson *et al.* (2015). For these, the same marginal cost as the Swedish CO_2 tax was applied per kg CO_2e. In addition, marginal costs for individual GHGs were applied, based on different estimates from Marten *et al.* (2015) and Waldhoff *et al.* (2014).

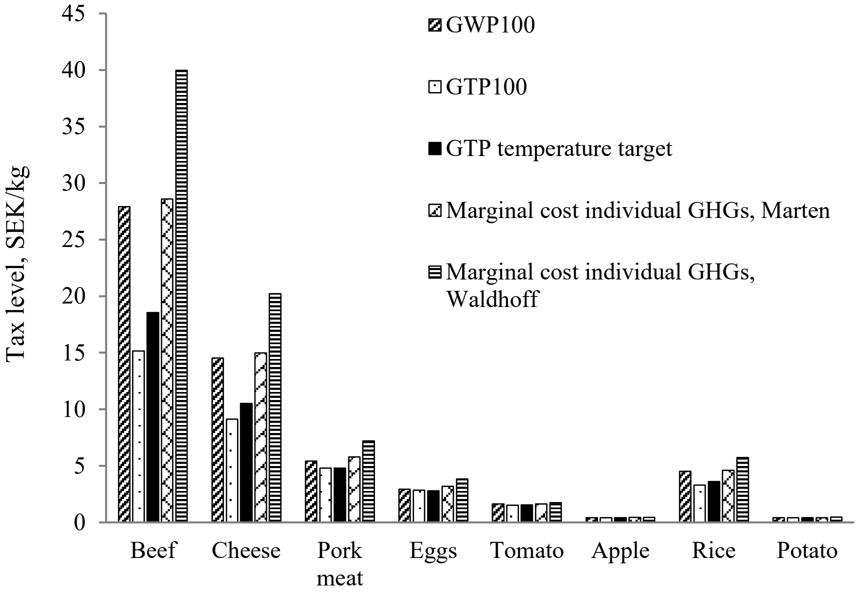


Figure 11. Tax rates (in SEK) for common foods on the Swedish market with different weighting of the climate impact and with different marginal costs applied.

In Papers I-II, the largest differences in tax levels were in general found for products where emissions are dominated by gases other than CO_2, especially CH_4, such as for beef, cheese and rice (Figure 11). This was due to variations in the weighting of their climate impacts, *e.g.* the weighting used in the GWP_{100} with climate-carbon feedbacks in Myhre *et al.* (2013) is 34 for CH_4 in relation to CO_2, compared with 11 using the GTP_{100} and 18 with the GTP factors suggested by Persson *et al.* (2015). A particularly large difference was observed between the GWP_{100} and the marginal damage cost according to Waldhoff *et al.* (2014), where the marginal damage cost of CH_4 was 45 times higher than that of CO_2. For products such as tomato, apple and potato, GHG emissions are dominated by CO_2, resulting in similar tax levels as the same marginal damage cost was applied (Figure 11). Notably, using the GWP_{100} and the marginal damage cost estimations according to Marten *et al.* (2015) gave similar results for all foods, which is explained by the similar weightings for different GHGs (32 for CH_4 and 391 for N_2O).

As discussed in Papers I-II, the fixed time horizon of 100 years for GWP and GTP is an arbitrary choice and can affect climate mitigation priorities, so using GTP factors as suggested by Persson *et al.* (2015) could be a better option for a tax designed to move consumption towards an actual climate goal. Implementing a tax based on the marginal damage cost of each individual GHG could be argued to be more cost-efficient, but there are large differences in estimates of marginal damage costs and the relations between CH_4 and N_2O. Instead, as concluded in Papers I-II, using the GWP_{100} as in current climate policy could be considered more suitable for consistency.

5.1.6 Other methodological choices to establish climate impact values

Other methodological choices in the modelling work performed in this thesis included choice of functional unit, as well as accounting for emissions and/or sequestration of CO_2 due to soil carbon changes and LUC (see section 4.1.2).

The climate impact was assessed using the functional unit per kg or litre of food, since this was the only viable choice to relate to the marginal damage cost of the emissions. If future developments allow for estimation of the marginal damage cost of a combined index of climate impact and nutrition, such an index could be argued to be attractive if it could steer towards simultaneous reduced climate impact and improved health of the population. However, it might be complex to estimate such combined costs, so targeting each issue by specific policies could be argued to be more cost-efficient (e.g. von Below *et al.*, 2017). Following from this, the climate impact of foods would then be targeted by a climate tax on foods, while additional health-related taxes could be implemented on unwanted individual components (*e.g.* sugar, saturated fat and sodium) and products (*e.g.* sugary drinks).

Emissions and/or sequestration of CO_2 due to soil carbon changes and LUC were included in the assessment of climate impact values. The choice of method to account for soil carbon changes was included in a sensitivity analysis in Paper II, which revealed differences depending on the method. It was found that the Tier 1 approach developed by the IPCC (2006)[3] could be used consistently across foods globally, but it did not capture the soil carbon sequestration potential of grass-clover leys (e.g. Poeplau *et al.*, 2015). The results using the ICBM approach were found to be in line with empirical data but, as pointed out in Paper II, this modelling might not correctly estimate the potential for imported products. However, research on methods to account for emissions and/or sequestration of CO_2 due to soil carbon changes and LUC is rapidly evolving, and future developments should be considered in the design of climate taxation.

[3]In this thesis, soil carbon changes were modelled using the Tier 1 approach following the guidelines by the IPCC from 2006. The guidelines were updated in 2019.

5.1.7 Using other climate impact datasets or standardisation to establish datasets

None of the existing datasets on the climate impact of food available at the time of the work described in this thesis was considered suitable for use in taxation (see section 3.4), or to study how methodological choices affect climate impact results. However, methods for calculating climate impact values are rapidly evolving, with multiple actors continually developing and updating the datasets. This thesis provides examples of how the climate impact of foods can be established and highlights how methodological choices affect the results, but other datasets on the climate impact could be suitable for use in climate taxation.

Likewise, using an alternative standardisation approach than that in ISO (2018) for establishing climate impact values might be suitable. For example, using the PCR for foods developed within the PEF framework (European Commission, 2018) or the EPD system (The International EPD System, n.d.) could be feasible if such standardisation is provided for all food products to be included in taxation.

5.2 Environmental sustainability of the Swedish diet in a local context

The aspects and indicators identified as important for assessing the environmental sustainability of food consumption in a Swedish context are shown in section 5.2.1. The results of calculations on the environmental pressures of foods are presented in section 5.2.2, while section 5.2.3 presents the results of benchmarking the environmental sustainability of the Swedish diet against global boundaries.

5.2.1 Environmental aspects and indicators for evaluating the environmental sustainability in a local context

Comparison of the indicators in the SEO and the EAT-*Lancet* frameworks (Paper III) revealed that several of the objectives could be linked to the variables in the EAT-*Lancet* framework. However, the focus of the EAT-*Lancet* framework is at a global level, so the variables need to be complemented by aspects and indicators brought forward in the SEO framework to capture environmental sustainability aspects at regional level in Sweden. Results from the comparison between the frameworks, current limitations and suggested indicators to use in identification of trade-offs from climate taxation are presented in Table 3 and are discussed in the following sections (based on the Earth system processes in the EAT-*Lancet* framework).

Climate change

With regards to the Earth system process of *climate change,* both frameworks were found to include indicators measuring GHG emissions aimed at limiting global warming. Since emissions of GHGs cause global problems regardless of the source of emissions, using a global perspective is suitable.

Land-system change

Regarding the Earth system process of *land-system change*, the EAT-*Lancet* indicator on cropland use focuses on limiting further expansion of agricultural land globally, as land clearance for agricultural land is currently a major driver of deforestation. The analysis in Paper III revealed that indicators related to land-system change in the SEO framework aim at maintaining current Swedish agricultural land by *e.g.* continued grazing by animals to preserve biodiversity-rich semi-natural pastures. Therefore, as an addition to an indicator on cropland use, an indicator focusing on pasture use was suggested to be included in evaluation of the environmental sustainability of the Swedish diet. The analysis in Paper III also showed that the indicators related to land use in the SEO include quality aspects of land use that are important for productivity, such as soil fertility and drainage, which were suggested for use as a complementary indicator of cropland use.

Nutrient cycling, freshwater use and biodiversity loss

The EAT-*Lancet* Earth system processes *nitrogen cycling* and *phosphorus cycling* aim at limiting addition of new reactive nitrogen and phosphorus to avoid eutrophication of terrestrial and marine ecosystems globally. As such, these indicators serve as proxies of the risk of eutrophication due to pressures of added nutrients, but do not take into consideration factors such as emissions intensities to specific catchments or the nutritional status of the recipient to which the nutrients are added. Similarly, the indicator of consumptive water use suggested in the EAT-*Lancet* framework (to assess the Earth system process of *freshwater use*) is a pressure indicator but does not analyse site-dependent impacts related to the use, such as water scarcity. Analysing freshwater use with a volumetric measure without considering site-specific impacts has been criticised within the LCA community. Researchers within the Water Footprint community have countered this criticism by arguing that water is both a local and a global resource, as it is traded worldwide via goods and products (Gerbens-Leenes *et al.*, 2021).

As discussed in section 3.3, site-dependent models can be used to assess impacts of emissions and resource use. In this thesis, a country-specific impact assessment was made to evaluate the Earth system process *biodiversity loss* from land use. However, biodiversity loss is primarily manifested on a regional level and should therefore ideally be assessed on a more detailed level. Similarly, for other site-dependent environmental aspects such as eutrophication and water scarcity, impacts may vary substantially within a country, so assessing the impacts at national level might not add to the representativeness of the results.

However, assessment of the site-specific environmental impacts of Swedish food consumption is currently inhibited by factors such as lack of detailed inventory data on *e.g.* emission intensities and resource use. Further, although it might be possible to assess the impacts of Swedish agriculture in *e.g.* eutrophication, it would be complex to link impacts to specific foods and diets. Such an analysis would potentially be even more complex for imported products, with lack of site-specific data and limited traceability of the origin of food imports to the Swedish market or of foodstuffs used as ingredients in prepared foods.

Due to the limitations identified with regard to using impact-orientated indicators, the indicators suggested by the EAT-*Lancet* Commission for nitrogen and phosphorus application, consumptive water use and terrestrial biodiversity loss were used in assessing the environmental sustainability of the Swedish diet and in evaluating taxation in Paper IV.

Additional aspects not captured by the EAT-Lancet framework

The analysis in Paper III revealed that several aspects related to food production and consumption in the SEO are not covered by the EAT-*Lancet* framework, including biodiversity loss of marine species, air pollution, acidification of freshwater and land, chemical pollution and ozone depletion. Additional indicators to capture these aspects should therefore ideally be included when assessing the environmental sustainability of the Swedish diet. Many of these aspects are included in the Earth system processes identified in the Planetary Boundaries framework (Steffen *et al.*, 2015; Rockström *et al.*, 2009).

Concerning marine biodiversity, it would be important to include an additional indicator of the status of fish stocks in Swedish and international waters. However, it was not considered feasible to include such a variable in the evaluation of taxation in Paper IV, due to lack of consumption data on fish and seafood for use in the demand described in Säll *et al.* (2020). Limitations identified for other environmental aspects included lack of inventory data on pesticide use in order to assess emissions of toxic substances to the environment. Data on pesticide use are available only on aggregated level for Sweden (Statistics Sweden, 2018), while data for other production countries within Europe are older and on an even more aggregated level (Eurostat, 2007). Further, no statistics on pesticide use are available for countries outside Europe, so data must be compiled from databases (*e.g.* Ecoinvent Centre, 2020) or inventories in previous studies.

Additional boundaries for evaluation of the environmental sustainability of Swedish food consumption

Boundaries specific for food production and consumption are not included in the SEO framework, and therefore benchmarking of the environmental sustainability related to boundaries other than those suggested in the EAT-*Lancet* framework is not possible. Defining boundaries for indicators from a local perspective and boundaries for the additional indicators identified in this thesis is therefore an important topic for future research. However, as the SEO framework primarily includes production-based indicators, additional boundaries would likely focus on targets for production. One exception is climate impact, where one of the indicators in the SEO focuses on consumption-based GHG emissions in Sweden and other countries. It has been suggested that specific targets for Swedish consumption-based emissions should be included in the SEO framework (Government of Sweden, 2020), which might include specific targets for Swedish food consumption.

Table 3. *Summary of similarities and differences between the SEO and EAT-Lancet frameworks, identified need for additional data or method developments and suggested indicators to use in evaluation of taxation*

Environmental category[1]	Aspects in the SEO not covered by the EAT-*Lancet* framework	Need for additional data or method development	Suggested indicator to use in evaluation of taxation
Climate change	-	-	GHG emissions
Land-system change	Maintain Swedish agricultural land, quality aspects of land use	System for monitoring soil fertility that can be connected to foods	Cropland use
	Maintain Swedish pasture	Improved statistics on different land types and uses of pasture	Pasture use
Nitrogen and phosphorus cycling	Site-dependent eutrophication impacts due to N application	Data on emission intensities for specific catchments and nutrient status of recipients to assess site-dependent eutrophication impacts	N application
	Site-dependent eutrophication impacts due to P application	Same as for N application	P application
Freshwater use	Site-dependent impacts of consumptive water use	Data on consumptive water use and availability on catchment level to assess site-dependent consumptive water impacts	Consumptive water use
Biodiversity loss	Local aspects of terrestrial biodiversity	Include state of threatened species regionally	Terrestrial extinction rate
	Marine biodiversity	-	-
Air pollution	Emission of air pollutants	-	NO_x emissions
Acidification of freshwater and land	Emission of acidifying substances	-	NH_3 emissions
Chemical pollution	Emission of toxic substances	Data on type and amount of pesticides used for different crops, to assess site-dependent impacts	Pesticide use (amount of active ingredient)
Ozone depletion	Emission of ozone depleting substances	-	N_2O emissions

[1]For the categories climate and land-system change, nitrogen and phosphorus cycling, freshwater use and biodiversity loss, these relate to the Earth system processes by the EAT-*Lancet* framework.

5.2.2 Environmental pressures associated with Swedish food consumption

The overall environmental pressures associated with the food available for consumption in Sweden and the contribution of different food groups to these pressures are shown in Figure 12. The values shown were calculated using the consumption data in Paper IV (see section 4.4). The burden per kg of different foods and food groups in the diet is shown in Figure 13.

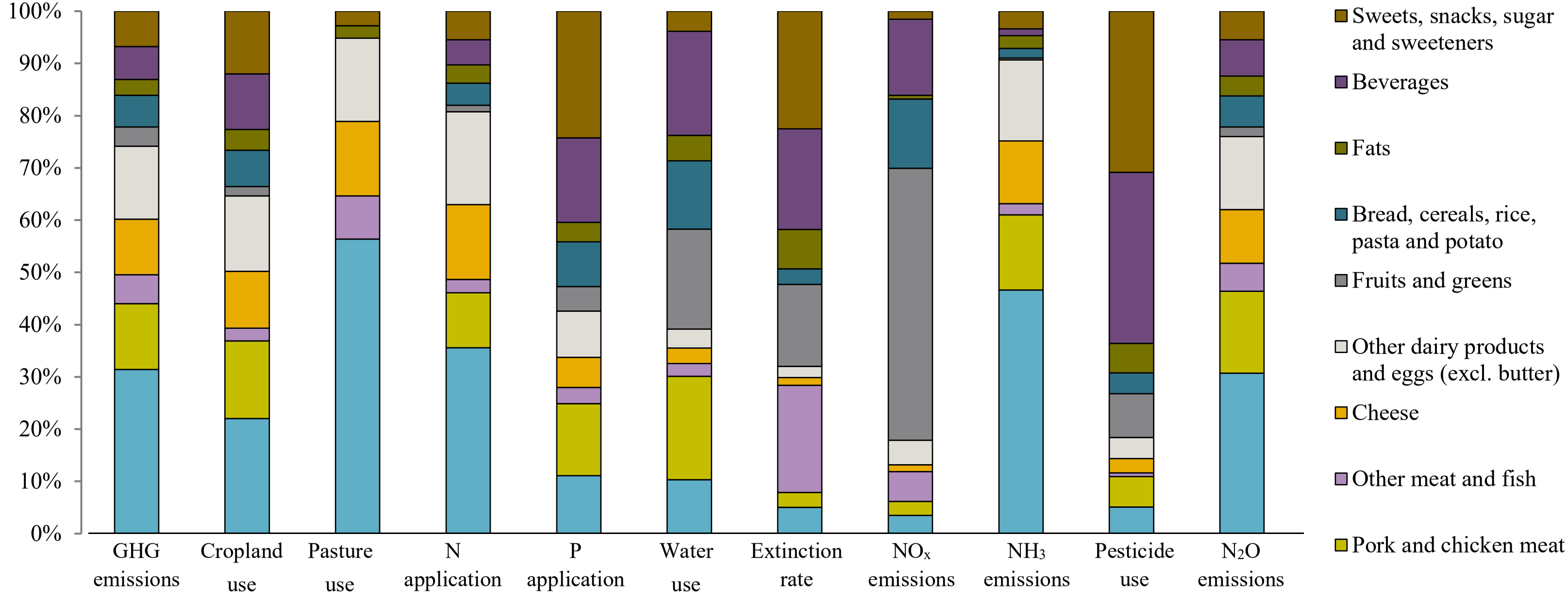
In terms of the overall pressures, many animal-based products such as beef, pork and chicken meat, cheese and other dairy products contributed a large shares of overall GHG emissions, N_2O emissions, cropland and pasture use, nitrogen application and emissions of NH_3 (Figure 12). This was because of the high burden per kg for these products, with especially pronounced pressures for *e.g.* beef, sheep meat and cheese (Figure 13). Other products with a high identified pressure per kg related to these categories included butter, cocoa and coffee.

With regard to phosphorus application, extinction rate and pesticide use, large shares of the overall pressures were found to be linked to consumption of beverages and sweets, mainly explained by the high pressure per kg caused by coffee and cocoa. Other products with high pesticide use and biodiversity impact per kg included olive oil and tropical fruits such as bananas. For coffee, cocoa and olive oil, the high biodiversity impact per kg was mainly explained by high cropland use. For other products such as bananas, the occupation of land for production in South and Central America caused high impacts due to high biodiversity loss per occupied m^2.

In comparison, animal products such as beef caused low biodiversity impacts per kg despite high land use, as most livestock production for the Swedish market takes place on relatively biodiversity-poor land in Sweden and Northern Europe. However, the impacts on biodiversity loss would change considerably if production were to take place in countries where the occupation of land causes higher biodiversity loss per occupied m^2. This was seen for sheep meat, where high land use together with high biodiversity loss from occupation of land for sheep production in New Zealand caused the highest biodiversity impacts per kg of all products studied.

With regard to freshwater consumption, an important share of the relative contribution was made by fruit and vegetables. This was explained by the large proportion of fruit imported from areas where high irrigation levels are often required. On a per kg basis, freshwater consumption was found to be especially high for rice, olive oil and coffee, which was also evident in the relative contribution.

For NO_x emissions (measured as emissions from transportation), the largest shares of the pressures were found for fruits such as bananas and oranges, as well as for coffee and rice. This was explained by the large transportation distances for these products, which are mainly imported from South and Central America and Asia.



19 Mton CO_2e	2.9 Mha	2.5 Mha	518 kton	42 kton	353 Mm^3	0.065 E/MSY	5.8 kton	63 kton	5.8 kton active ingred-ient	13 kton

Figure 12. Environmental pressures of Swedish food consumption for the whole population, and contribution of different food groups to the overall pressures calculated with consumption data used in Paper IV. Water use refers to consumptive water use.

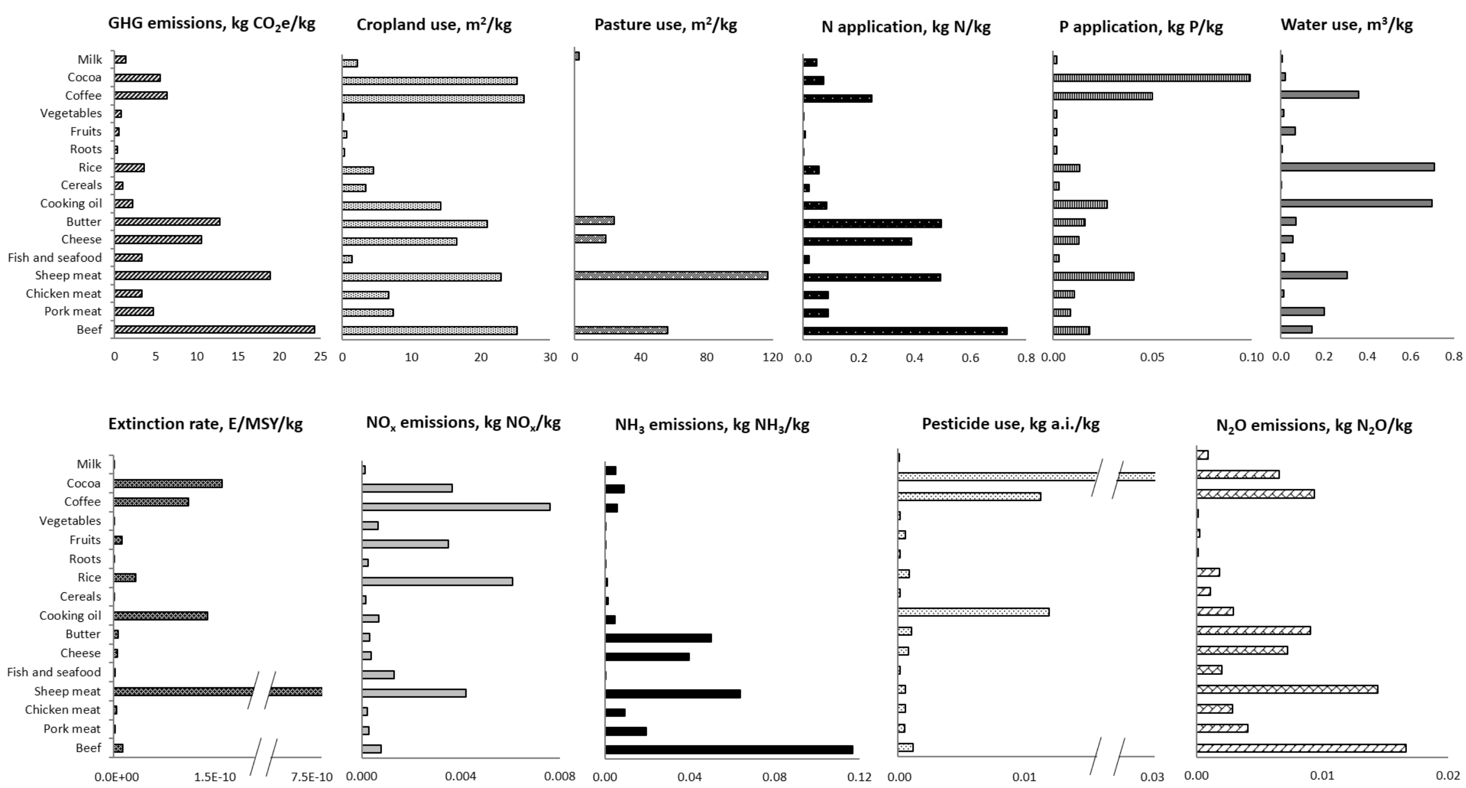


Figure 13. Weighted average environmental pressure per kg for foods available on the Swedish market. a.i. = active ingredient. Cheese is a weighted average of hard cheese and dessert cheese. Water use refers to consumptive water use.

5.2.3 Environmental sustainability of the Swedish diet

The results from Paper III regarding the environmental pressures associated with Swedish food consumption per capita, benchmarked against downscaled per-capita boundaries given in the EAT-*Lancet* framework, are illustrated in Figure 14 and presented in absolute numbers in Table 4. For pasture use, emissions of NO_x, NH_3 and N_2O and pesticide use, no boundaries have been set and these were not included in the benchmarking.

The analysis in Paper III revealed that current Swedish food consumption exceeded all environmental boundaries except freshwater use, where the diet was still well below the boundary (Figure 14). With regard to GHG emissions, it was found that the average Swedish diet exceeded the allowed boundary for overall emissions of GHGs by more than three-fold. Of the 2.2 ton CO_2e emitted per capita and year, emissions of CO_2 accounted for 0.92 ton (~41%), while emissions of CH_4 and N_2O together accounted for 1.3 ton CO_2e (~58% of total emissions). Emissions of HCFC-22 (0.01 ton CO_2e) made up a minor fraction (<1%).

To reach climate goals such as those in the Paris Agreement, reductions in CO_2 down to zero are needed, including negative emissions. Deep reductions in CH_4 emissions have also been pointed out as necessary (IPCC, 2018), but some emissions of CH_4 at a constant level may be permissible due to the short residence time of CH_4, as emissions will replace CH_4 that is broken down. With current emissions, however, both the target of net zero emissions of CO_2 and a maximum of 0.68 ton CO_2e for CH_4 and N_2O were transgressed (Paper III). Further, even if emissions of CO_2 were reduced to zero, the boundary would still be exceeded by almost two-fold due to emissions of CH_4 and N_2O.

With regard to cropland use, the results in Paper III showed that the Swedish diet required use of almost twice the cropland area set as the EAT-*Lancet* boundary. The results on GHG emissions and land use were similar to those reported previously by Röös *et al.* (2015), who found that the average Swedish diet exceeds the sustainable level of both climate impact (2.5-fold the limit) and land use (by ~1.1-fold the limit).

Concerning application of nutrients in agriculture, the boundaries for both nitrogen and phosphorus were found to be transgressed by four-fold. For rate of extinctions, the Swedish diet was found to cause six-fold more extinctions than the boundary. For all categories where the boundaries were transgressed, the pressure was well above the zones of uncertainty, with the exception of biodiversity (Table 4).

The global food consumption assessment by the EAT-*Lancet* Commission (Willett *et al.*, 2019; Springmann *et al.*, 2018) revealed similar trends as found in this thesis, *i.e.* with the safe operating spaces being exceeded for climate change, phosphorus cycling and biodiversity loss. With regard to global freshwater and cropland use, the pressures were within the boundary. However, for cropland use, due to the increasing world population, the boundary was projected to be transgressed by 2050 if measures to reduce waste, improve production or change diets are not imposed.

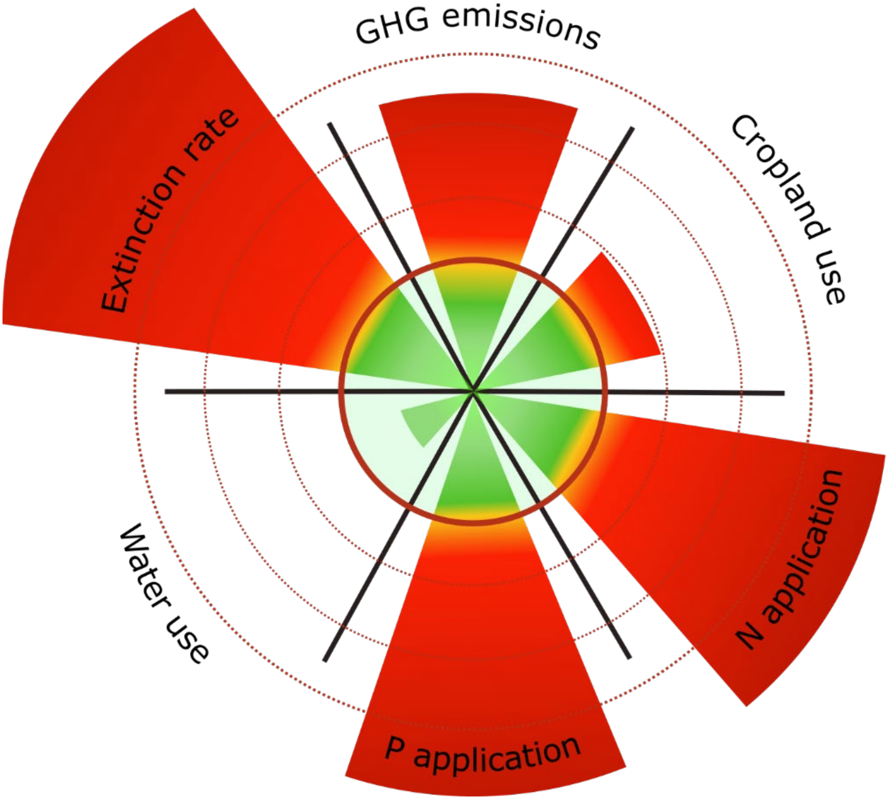


Figure 14. Results from Paper III on the environmental pressures associated with Swedish food consumption, benchmarked relative to the boundaries suggested by the EAT-*Lancet* Commission (Willett *et al.*, 2019). The red inner circle indicates the per-capita boundaries, *i.e.* 100% of the 'allowed' boundary, and each dotted outer circle shows exceedance of the boundary by 100%. Water use refers to consumptive water use.

Table 4. *Results from Paper III on the environmental pressures per capita associated with Swedish food consumption, benchmarked against downscaled per-capita boundaries for the control variables given in the EAT-Lancet framework. Range of uncertainty for the boundaries is given in brackets*

	GHG emissions	Cropland use	N application	P application	Consumptive water use	Extinction rate
Environmental pressures per capita	2.2 ton CO_2e per year (0.92 ton CO_2, 0.82 ton CH_4*, 0.5 ton N_2O*, 0.01 ton HCFC-22*)	0.34 ha	57 kg N per year	5.0 kg P per year	55 m^3 per year	8.3×10^{-9} E/MSY
Per capita boundary (downscaled from the global boundaries given by the EAT-*Lancet* Commission)	0.68 ton CO_2e per year for CH_4 and N_2O and zero for CO_2 from fossil fuels, land use and LUC (0.68–0.73)	0.18 ha (0.15–0.2)	12 kg N per year (8.8–18)	1.1 kg P per year (0.8–2.2)	339 m^3 per year (136–542)	1.4×10^{-9} E/MSY (1.4×10^{-10}–1.1×10^{-8})

*Expressed in CO_2e.

5.3 Environmental effects and potential goal conflicts of taxation

The environmental effects and potential goal conflicts of climate taxation for all food products on the Swedish market are presented in section 5.3.1, while section 5.3.2 presents the results for the alternative taxation scenarios. Section 5.3.3 discusses potential goal conflicts from taxation not captured by the modelling. In section 5.3.4, potential social and economic goal conflicts from climate taxation are discussed.

5.3.1 Environmental effects and potential goal conflicts of taxation of all foods on the market

Evaluation of the environmental effects of climate taxation in Paper IV revealed that it decreased the burden of all environmental categories by between 7 and 12 % (Table 5). The largest effects were seen for pasture use, with a reduction of 310 000 hectares of land, and for ammonia emissions, with a reduction of 7300 ton (both corresponding to a 12% decrease in current levels). The decline in environmental pressures was mainly explained by an overall reduction in food consumption, of about 7% of current amounts (in kg or litre). As discussed in section 4.4, this change could be a result of decreased actual consumption and decreased food waste. Products with the largest decrease in consumption included many animal products, such as milk, cheese, beef and chicken, which mainly was explained by their high climate impact resulting in a higher tax rate. Furthermore, a large decrease was seen for non-alcoholic beverages such as fizzy drinks and cider. The only products which were found to increase were sugar and sweeteners, but the increase was small.

The large decline in pasture area was mainly explained by decreases in consumption of beef. Due to limitations in calculation of consumer demand, which was used to assess the effects of taxation in Paper IV, it was not possible to determine whether the decline in beef consumption derived from meat produced in Sweden, or from animals grazing on cropland or semi-natural pastures. Hence, no conclusion could be drawn on whether a decrease in demand for land due to climate taxation would lead to positive or negative effects for biodiversity conservation, depending on the global or regional perspective of land use (see discussion in section 5.2.1).

In Paper IV, it was found that although a climate tax would lead to reductions in consumption of beef, the yearly per-capita consumption would still be around 22 kg beef. In comparison, a study by Röös *et al.* (2016) found that maintenance of the current area of Swedish semi-natural pastures could be compatible with reducing per-capita consumption of beef to 4-14 kg per year. Hence, as concluded in Paper IV, current consumption levels of beef could probably be reduced without creating a goal conflict with preservation of Swedish semi-natural pastures. Similarly, Larsson *et al.* (2020) concluded that there is no shortage of ruminant animals to maintain Swedish semi-natural pastures. However, due to the high cost to farmers of rearing their animals on such pastures in comparison with other production systems, animals graze on cropland or are housed for long periods of the year, including during the grazing season. According to Larsson *et al.* (2020), targeted policy instruments are needed for maintenance of semi-natural pastures, *e.g.* by increased payments to farmers for management of these areas using grazing animals. Such payments could potentially be supported using revenues resulting from climate taxation (Gren *et al.*, 2021).

Table 5. *Environmental effects of climate taxation for the whole Swedish population*

	Absolute effects of taxation for the Swedish population	Effects of taxation in %
GHG emissions	-1.9 Mton	-10%
Cropland use	-0.28 Mha	-9.7%
Pasture use	-0.31 Mha	-12%
Nitrogen application	-57 kton	-11%
Phosphorus application	-3.4 kton	-8.1%
Consumptive water use	-26 Mm^3	-7.5%
Terrestrial extinction rate	-0.0045 E/MSY	-7.0%
NO_x emissions	-0.43 kton	-7.4%
NH_3 emissions	-7.3 kton	-12%
Pesticide use	-0.45 kton active ingredient	-7.9%
N_2O emissions	-1.4 kton	-10%

5.3.2 Environmental effects and potential goal conflicts of alternative taxation scenarios

Figure 15 shows the environmental effects of the reference scenario involving targeting all foods on the market with a climate tax, compared with alternative tax scenarios modelled in Paper IV. The alternative scenarios included different sets of food products (only animal products, only beef or only monogastric meat and eggs) and changes to the VAT system (increasing rates of animal products to 25%, reducing rates on fruit, vegetables and cereals to 6%, or using a combination of both).

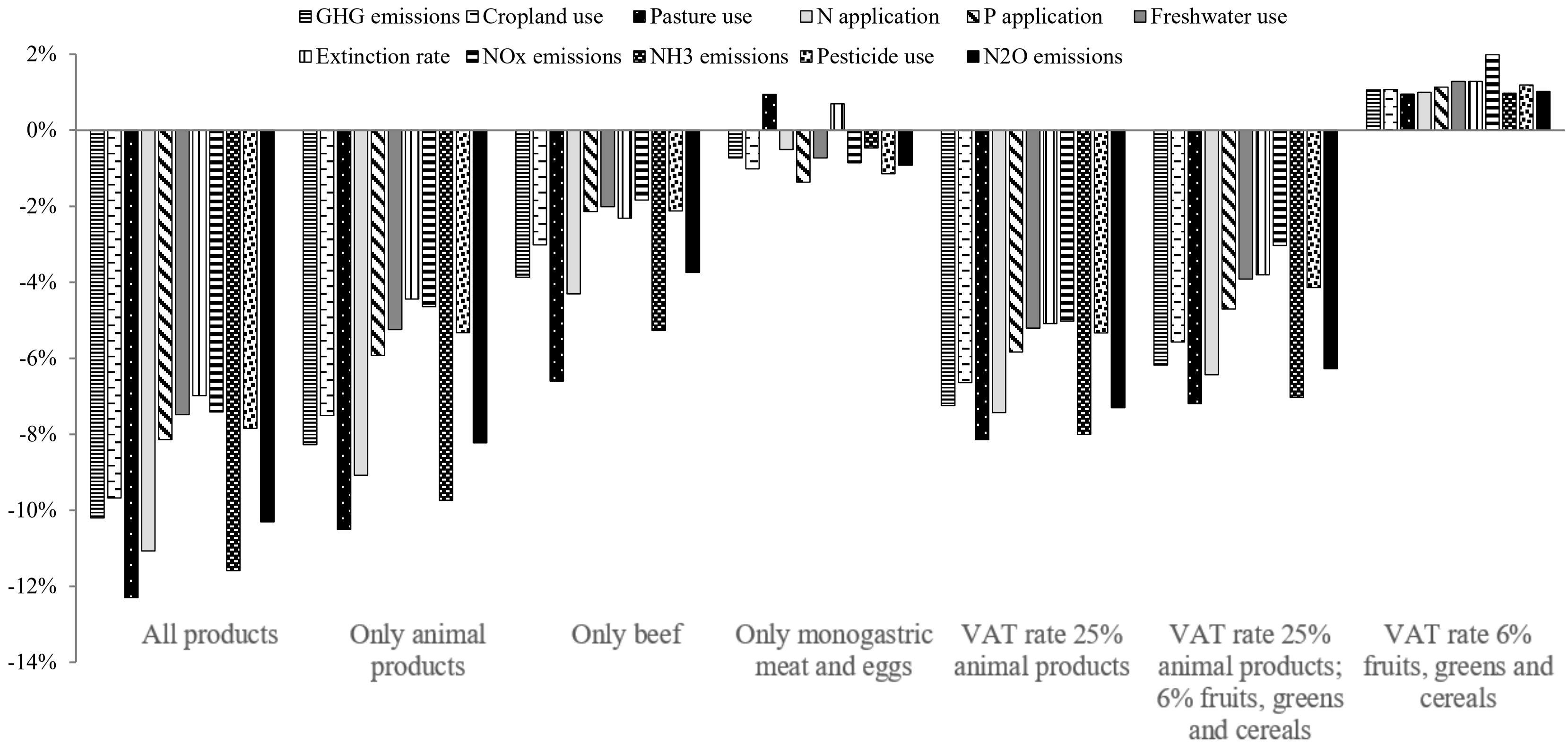


Figure 15. Results from Paper IV on the environmental effects of (left) taxation in the reference scenario of including all products and (right) alternative taxation scenarios involving different sets of food products and making changes to the VAT system.

Including only animal products and increasing the VAT rate on animal products to 25% resulted in similar effects in all environmental categories to the reference scenario of targeting all food products with a climate tax. Thus while a tax targeting all products decreased the climate impact by 10%, including only animal products or increasing the VAT rate on animal products decreased the climate impacts by 8.3% and 7.2%, respectively. The relative effects for climate impact and the other environmental categories were similar in the mentioned scenarios. As discussed in Paper IV, changing VAT rates would involve making changes to an existing tax system, which might ease the administration burden compared with implementing a new tax system, as would be the case with a climate tax. However, such a tax would not reflect the climate impact of the products and emissions would not be taxed similarly to other sources (as with the current taxation on CO_2 emissions), so a climate tax would be more cost-efficient than changing the VAT rates.

Reduced VAT rates on fruit, vegetables and cereals increased the burden in all environmental categories, owing to an overall increase in food consumption. Similar results were found in a previous study by Broeks *et al.* (2020), who pointed out that increased consumption of plant-based foods could still lead to a net societal benefit due to *e.g.* reduced healthcare costs. Using a simultaneous increase in VAT rates on animal-based foods and decrease in VAT rates on fruit and vegetables resulted in overall reductions in climate impact (by 6%) and other environmental aspects (by between 3.1% and 7.1%). Making such a simultaneous change could potentially lead to higher acceptance than targeting only animal-based products, as consumers would be compensated. For example, as discussed in section 3.2.3, Drews and Van den Bergh (2016) suggest that individuals to a larger extent tend to accept policies such as subsidies due to the lower perceived financial costs to the individual.

Finally, in the scenario where only monogastric meat and eggs were taxed, the effects on climate impact and on many other environmental categories were small (*e.g.* less than 1% decrease in current GHG emissions), but with an increase in both pasture use and extinction rate. This was mainly explained by a rise in consumption of other meat products and especially sheep meat, for which the results in Paper III indicated a high biodiversity impact for sheep meat from New Zealand (see Figure 13). However, these results are highly sensitive to the production region, as Paper III showed that assuming a scenario where all sheep meat was produced in Sweden would lead to decreased impacts on biodiversity. Further, as discussed in Paper IV, taxing only monogastric meat and eggs would exclude beef and sheep meat from taxation, which would probably be difficult to justify due to their high climate impact. However, the results give an important indication of how increased consumption of products from sensitive regions can exacerbate the burden on biodiversity (see also discussion in section 5.3.3).

Due to the overall decrease in pasture use in most scenarios, a potential goal conflict could arise if the decrease were accompanied by a decrease in beef and sheep meat from animals grazing on Swedish semi-natural pastures, as discussed in section 5.3.1.

5.3.3 Potential environmental goal conflicts not captured by the modelling

The level of detail in simulations of the environmental effects of climate taxation was limited by data availability on food consumption (section 4.4). Although overall food consumption was seen to decrease due to taxation, there could be changes within certain food groups that were not captured by the modelling and which could exacerbate pressures. For example, as discussed in Paper IV, if pork and chicken consumption from production using substantial amounts of soy were to increase, pesticide use could also increase. There could also be site-specific impacts from taxation which are not captured by current indicators for *e.g.* nutrient application and freshwater use, as discussed in section 5.2.1.

Plant-based substitutes for animal products, for which demand has increased in recent years (Zachrisson, 2019), were not included in the simulations. Further, the effects of fish and seafood were evaluated using an aggregated group for these products. Due to these limitations, no specific evaluation was made of the impacts on marine biodiversity. In the following sections, potential goal conflicts from these limitations are discussed.

Plant-based substitutes to animal products

The environmental impacts of plant-based alternatives to meat and dairy were not evaluated in this thesis, but have been included in other studies (e.g. Karlsson Potter *et al.*, 2020). Plant-based protein substitutes for meat include soya mince and tofu, which tend to have a lower climate impact than beef, pork and chicken meat. Concerning land and freshwater resource use, both beef and pork meat have a higher environmental burden than most protein-based alternatives to meat, while chicken meat is at a similar level to plant-based substitutes. The biodiversity impacts of plant-based substitutes are generally at the same level as those for pork and chicken meat. As such, an increase in demand for these products would probably not lead to increased environmental burden (Karlsson Potter *et al.*, 2020).

Plant-based substitutes to milk and cream are based on *e.g.* soy, oats, coconut and almond, while plant-based alternatives to cheese generally include coconut oil. Most plant-based drink alternatives have a lower climate impact, land use and biodiversity impact than milk. However, for coconut milk, which is used as a substitute for dairy cream, the biodiversity impacts are substantially higher. This is mainly explained by the high risk of biodiversity loss in production areas in *e.g.* the Philippines and Indonesia. The same applies to plant-based cheese, which often contains a high share of coconut oil and thus poses a risk of high environmental burden (Karlsson Potter *et al.*, 2020).

Thus while increased consumption of plant-based products in general could decrease climate impacts, there could be a risk of exacerbated environmental burdens with regard to biodiversity if animal-based products were replaced by these products following climate taxation.

Fish and seafood

Simulations on the environmental effects of taxation revealed an overall decrease in consumption of fish and seafood in most scenarios. Exceptions were seen *e.g.* in the scenario targeting only beef, with a slight increase of 0.33% on current consumption levels. No evident goal conflict was identified between the studied environmental pressures through these changes. However, there is a potential risk of the pressure on marine biodiversity being exacerbated by this increase, or by sustained consumption of fish and seafood involving vulnerable fish stocks and/or species from sensitive regions.

Marine biodiversity has been included in LCA and environmental sustainability assessments through methods taking into consideration variables such as overfishing (Emanuelsson *et al.*, 2014) and biomass removal (Hélias *et al.*, 2018). Further, in the WWF's consumer 'Fish Guide' (WWF, n.d.), the marine biodiversity of fish species is evaluated qualitatively based on the impact of fisheries on the state of fish stocks and ecosystems. Depending on the fishing region and gear type used by fishing fleets, the WWF guide identifies several of the most common fish species in Sweden as potentially hazardous for marine biodiversity, including cod, Alaska pollock, European plaice and hoki. For these, the Fish Guide recommends consumption only of fish certified by the Marine Stewardship Council (MSC) or Swedish KRAV. In general, fish species caught using bottom trawling have negative impacts on ecosystems (WWF, n.d.), including cod, saithe and northern prawn. Further, Emanuelsson *et al.* (2014) found that the majority of European commercial fish stocks are overexploited to varying degrees, including stocks of cod, European plaice and herring.

In summary, while no evident goal conflict was identified specifically for consumption changes related to the category of fish and seafood, consumption of certain species could lead to exacerbation of marine biodiversity loss.

5.3.4 Potential social and economic goal conflicts

Apart from the potential goal conflicts between different environmental aspects, social and economic goal conflicts could also arise from taxation. These could include *e.g.* effects on nutrient intake and distributional effects on the Swedish population, as well as effects on the income of Swedish farmers.

In Paper IV, the energy intake of the Swedish population following taxation was estimated using the results on food consumption changes. If the decrease in food consumption stemmed from reduced food waste, there would naturally be no change in the energy intake. If the change were to be caused by an overall decrease in actual consumption, the results indicated that a climate tax targeting all foods on the market could reduce current energy intake by up to 172 kcal per capita and day. Current intake from Swedish food consumption is approximately 2800 kcal per capita and day at population level, while the average recommended intake is between 1700 and 3200 kcal per day, depending on age, sex and level of physical activity (Swedish National Food Agency, 2020). Compared with the reference level, calorie intake after taxation was found to decrease by about 6% without leading to insufficient recommended energy intake. However, it should be emphasised that the analysis was simplified, as it only included average caloric intake at population level.

Other nutrients were not studied in this thesis, but were assessed in the overall project within which the thesis work was performed (Röös *et al.*, 2021). Due to the overall decrease in food consumption, a general decline was seen in the intake of nutrients such as proteins, vitamin B12, vitamin D, folate and iron. Concerning protein and vitamin B12, current consumption levels are above the recommended levels, so a climate tax would probably not lead to deficiencies in intake. However, with regard to vitamin D, folate and iron, current levels are too low compared with the recommendations, so a climate tax could risk exacerbating the already deficient nutrient intake. Part of the potential goal conflict could be counterbalanced by replacing meat with legumes (Röös *et al.*, 2018).

Concerning distributional effects from a climate tax, Säll (2021) found that a climate tax would be regressive, *i.e.* that low-income earners would pay a larger share of their income on the climate tax. The effect would be largest for families with several children, pensioners and the unemployed, who devote a larger share of their expenditure to food than other consumers and households.

To balance such goal conflicts, Säll (2021) pointed out the necessity of implementing policies to compensate consumers when introducing a climate tax. For example, compensation could be made using the revenues resulting from climate taxation. As discussed in section 3.2.3, acceptance of policy could increase if richer members of society paid a larger share and potential revenues were recycled to poorer or more vulnerable groups in society (Drews & Van den Bergh, 2016).

Similarly to using the tax revenue to compensate consumers, compensation could also be paid to Swedish farmers negatively affected by a climate tax. Apart from the previously mentioned subsidy for maintenance of semi-natural pastures (section 5.3.1), Gren *et al.* (2021) investigated the potential for compensating farmers for supporting ecosystem services for additional emission reductions, such as restoration of drained peatlands (see section 5.4 for a discussion on the emissions from drained peatlands) or establishment of wetlands to reduce nutrient leaching. The findings by Gren *et al.* (2021) indicate that the emissions reductions from restoration of drained peatlands could even exceed those achieved by changes in consumption following a climate tax. Whether such compensation measures would be compatible with Swedish legislation or the EU Common Agricultural Policy would need further evaluation.

5.4 Uncertainties and limitations

5.4.1 Inventory data and models used

For some variables used in the modelling, such as yield levels, inventory data are readily available in national and global databases (FAO, 2021; SBA, 2021). Lack of inventory data or uncertainty in the data can be a problem for other variables, for Sweden but especially for countries outside Europe. For example, data on fertiliser application rates are available through statistical databases and advisory services in Sweden and other European countries (*e.g.* Statistics Sweden, 2021), but recent data are lacking for certain crops or production countries. With regard to water use, inventory data are available in the WaterStat database (Mekonnen & Hoekstra, 2012; Mekonnen & Hoekstra, 2011). The values do not represent actual water use for irrigation, but are the result of modelling of crop water requirements. Mekonnen and Hoekstra (2011) used irrigation maps to deduce which crops are irrigated, but point out that such maps only are available for a limited number of crops and certain crops may be assumed not to be irrigated. Using such data may therefore lead to underestimation of consumptive water use associated with the Swedish diet. On the other hand, for crops assumed to be irrigated, consumptive water use was assumed to equal irrigation demand, which may overestimate actual irrigation levels as optimal levels may not always be met in regions with scarce water availability.

Many of the emission sources in this thesis were calculated using simplified models. For example, N_2O emissions from managed soils were estimated using default emission factors set by the IPCC, which assumes that a fixed fraction of the nitrogen applied to mineral soils is emitted as N_2O (IPCC, 2006). In the 2019 refinement to the 2006 guidelines by the IPCC, the default factors were disaggregated for wet and dry climates, and depending on whether the nitrogen input derives from synthetic fertiliser or other nitrogen source (IPCC, 2019b). Although using the more disaggregated values reduces uncertainty, the method still provides coarse estimates of the emissions. Other methods to account for soil emissions could enable more detailed differentiation of soil emissions, give better agreement with measured emissions and potentially reduce uncertainty (Henryson, 2019), but would require better traceability of products, as discussed in section 5.2.1.

5.4.2 Benchmarking environmental pressures relative to global boundaries

The modelling of terrestrial extinction rate required a choice of amortisation period for the overall potential species loss, but the choice was found to have a strong impact on the results. The species losses were allocated over a time horizon of 100 years, which is in line with the time period used for the climate metric applied in the thesis (GWP_{100}). Similarly to the discussion in relation to climate metrics (section 5.1), other time horizons are also possible for amortisation of extinctions. According to Paper III, allocating all of the impacts to the same year would, naturally, lead to a 100 times larger impact, which would be 600-fold the EAT-*Lancet* boundary. Using a shorter time horizon of 20 years would give impacts of 30 times the boundary, while allocating the species loss over 500 years would cause impacts of 1.2-fold the boundary.

Setting global boundaries for the food system is highly challenging, since the drivers of Earth system processes are complex and interconnected (Willett *et al.*, 2019). The EAT-*Lancet* boundaries for GHG emissions and nitrogen application have been criticised for being based on the unavoidable share of emissions and resources needed to feed the global population, rather than on absolute biophysical limits for Earth systems within which humanity should operate. This is discussed by Einarsson *et al.* (2019), who point out that the boundaries should rely upon scientific evidence on the limits of the Earth systems in order to be scientifically consistent, although this causes trade-offs between reaching environmental targets and maintaining current levels of prosperity.

In order to benchmark the environmental impacts of the per-capita Swedish diet against the global EAT-*Lancet* boundaries, the boundaries were downscaled to equal per-capita boundaries for the global population. This approach enabled a straightforward and simple illustration of the contribution of the average Swede and the global citizen to globally 'allowed' emissions and resource use from the food system, regardless of where the pressures were caused. Several other methods could be used to allocate the emissions and resource space of the global boundaries, *e.g.* based on perspectives of equity of factors such as historical emissions or resource use. With such an approach, less developed countries could be allowed higher levels of emissions or resource extraction, based on their lower contribution to the problem historically and on their ability to pay (Baer *et al.*, 2009). Further, downscaling the boundaries could be done according to the spatial resource availability within a country or region, *e.g.* based on land and water resources (Fang *et al.*, 2015).

5.4.3 Excluded emission sources

The method devised in Paper II was developed with the aim of establishing climate impact values for use in taxation, so emission sources of minor importance for the climate impact per kg of food were excluded (section 4.1). Such emissions sources were *e.g.* production of seeds and pesticides and energy use for storage at wholesaler and retailer level. Another emissions source which was not included in the climate impact assessment in Paper II was cultivated organic soils. Emissions from such soils have been estimated to account for 6-8% of total annual anthropogenic GHG emissions in Sweden (Berglund & Berglund, 2010). About 7% of Swedish agriculture is based on organic soils, of which half is arable land (Pahkakangas *et al.*, 2016). Organic soils were originally pristine peatlands which were drained for agricultural purposes in past centuries. These soils are rich in organic matter and, following drainage, the material becomes aerated and starts to decompose, resulting in emissions of GHGs. Based on the scientific literature at the time of writing this thesis, no specific cropping system on cultivated organic soils can limit the GHG emissions (Norberg, 2017). Further, since the emissions would occur even if soils were not cultivated, it would be difficult to allocate them to specific crops in order to include them in climate taxation. Another option could be to allocate the emissions equally to all foods produced on organic soils in Sweden, or to all foods produced in Sweden based on the land use per kg food. However, targeting emissions from organic soils by a consumption tax on food would be an indirect policy instrument which would not target the actual emissions, as consumption choices have not been shown to influence the emissions from cultivated peat soils. Rather, policy instruments on the production side have been indicated as a potentially cost-efficient policy option by the Swedish government and will be included in a future subsidy system for management of organic soils (Government of Sweden, 2021).

In assessing the environmental sustainability of the Swedish diet in Paper III, the method from Paper II was applied and thus the assessment excluded the above-mentioned emissions. Including these emissions would have led to even greater overshooting of the environmental boundary relating to GHG emissions.

5.4.4 Concluding remarks on estimating uncertainties

As discussed, uncertainties in the calculations performed within this thesis are due to various aspects such as data and model uncertainty as well as choice of methodology. Analysing uncertainties can help to improve the robustness of results, which is important when using LCA results for implementation in policy (e.g. Sala *et al.*, 2016). In this thesis, the influence of methodological choices on climate impact values was assessed through various sensitivity analyses. As pointed out by Björklund (2002), methodological choices such as those discussed in this thesis may override many other types of uncertainty, and therefore serve as an important tool to illustrate the influence on the results.

With regard to uncertainties associated with inventory data and the models used, making statistical estimates of these uncertainties would be associated with major difficulties due to *e.g.* data limitations on variations in input data. For environmental indicators assessed in this thesis, such as cropland use, the calculations are straight-forward as they are based on crop productivity levels, calculated from yield statistics. For GHG emissions, important emissions arise in several process steps in the production chain and in many of these processes emissions are variable due to *e.g.* climate, soil conditions and management practices. While it might be possible to provide gross uncertainty ranges for *e.g.* cropland use by using data on variability in crop productivity levels, it would be a complex task to establish statistical uncertainties for indicators such as GHG emissions for all food products included in this thesis and for whole diets. Establishing uncertainty ranges is an important topic for further studies, for example when optimising diets to fit within estimated environmental boundaries.

5.5 Applying food taxation and other policy approaches to reduce environmental impacts from the food system

Although climate taxation and consumer price changes by differentiating VAT rates offer potential for decreasing the climate impact and other environmental pressures, it is clear that the pressures would be high even after taxation. Therefore, as discussed in Paper IV, taxation of food cannot be used as a stand-alone policy for curbing environmental impacts from food consumption. Instead, as previously mentioned, both production- and consumption-side measures are necessary to achieve profound changes in the food system. Food taxation could be one of several public policies implemented in a policy package.

Other approaches to steer towards consumption-side changes and achieve dietary changes could include informative policies such as awareness raising, 'negative' labelling of environmental impacts and 'nudging'. Using informative policies simultaneously with introducing a climate tax might also increase acceptance of the tax. Apart from policies directed towards consumption, a policy package should also be directed towards the supply chain and include food industry and retail (Röös *et al.*, 2020). For example, the retail sector could be required to report and improve on a set of key performance indicators related to the environmental impacts of food sold, which is already used in other sectors, *e.g.* in sales of new cars in the European Union, for which regulations are set on a maximum of GHG emissions per km (EU, 2019).

Ultimately, changing consumption patterns to environmentally sustainable diets will require changes in norms and habits, where policy can be a factor supporting such changes (Nyborg *et al.*, 2016). For example, anti-smoking laws in public areas in Norway have contributed to new norms on not smoking indoors (Nyborg & Rege, 2003). However, the effects of policy on norms also depend on the context, since the smoking norm in *e.g.* Greece prevails despite anti-smoking laws (Nyborg *et al.*, 2016). As discussed in section 3.2.3 of this thesis, policy support is affected by a range of aspects, including contextual factors, with increased support for policy seen in countries with a general trust in *e.g.* society and politicians (Drews & Van den Bergh, 2016). Aspects considering acceptance and changes in norms could therefore be included in future studies on food taxation.

6 Conclusions

- Consistent and transparent datasets on the climate impact of foods, to be used in a climate tax on Swedish food consumption, were established using a simplified method based on LCA methodology.
- Evaluation of methodological choices for establishing the climate impact values revealed that the final results might not (in the short term) reflect what is theoretically cost-efficient, as the choice of method also needs to consider factors such as simplicity in calculations and maintenance.
- For the choice of system boundary, adjusting for existing taxation would theoretically achieve a cost-efficient tax, but would probably lead to a high administration burden. Including only biological GHG emissions would be a simpler alternative to target currently untaxed emissions, but this would involve a risk of fossil CO_2 emissions remaining untaxed if imports were made from countries where CO_2 taxes are not in place.
- For the choice of weighting emissions and tax levels of different GHGs, taxing each gas individually would theoretically be most cost-efficient, but estimates of the marginal damage costs of GHGs vary. Using GWP_{100} could be an advantage for consistency, as it is used in current climate policy.
- The global EAT-*Lancet* variables capture several important aspects relevant for environmental sustainability concerns of Swedish food consumption, but lack indicators to capture aspects such as pasture use, marine biodiversity, air pollution, acidification, emissions of toxic substances and ozone-depleting substances.
- Using the indicators in the EAT-*Lancet* framework enables the environmental pressures of emissions and resource use related to food consumption to be assessed. To perform assessments at finer resolution and to increase the representativeness of potential impacts, site-dependent impact models should ideally be used.

- To enable inclusion of complementary indicators that better capture the environmental sustainability of the Swedish diet, there is a need for better inventory data on emission intensities (*e.g.* for nutrients to assess eutrophication impacts) and resource use (*e.g.* for pesticide use), together with better traceability data for foods imported to the Swedish market.
- The environmental pressures of Swedish food consumption were found to exceed global environmental targets for GHG emissions, cropland use and application of nitrogen and phosphorus by two- to four-fold. For extinction rate, the boundary was exceeded by six-fold. The only environmental category for which the global target was not transgressed was freshwater use, where the pressure of the diet was well below the limit.
- Meat and dairy products were found to make large contributions to GHG emissions, land use, application of nitrogen and emissions of NH_3.
- With regard to phosphorus application, extinction rate and pesticide use, large shares of the overall pressures were linked to consumption of coffee and cocoa, due to their high impact per kg. Other products with a high biodiversity impact and high pesticide use per kg included olive oil. Especially high biodiversity impact was seen for sheep meat, due to high land use together with high biodiversity loss from occupation of land for sheep production in New Zealand.
- An important share of the relative contribution to freshwater consumption was made by fruit and vegetables, whereas freshwater use on a per kg basis was especially high for rice, olive oil and coffee.
- Climate taxation of all foods on the Swedish market was found to have potential to decrease food-related environmental burdens by 7-12%, including aspects such as cropland use, nutrient application and ammonia emissions. The effects were mainly explained by the overall decrease in food consumption.
- Due to the relatively large decline in beef consumption, pasture use was found to decrease by up to 12%. This is positive from a global perspective by limiting further expansion of agricultural land. From a Swedish perspective, however, reducing consumption of beef could potentially create a goal conflict with maintaining biodiversity in semi-natural pastures. To avoid such goal conflicts, increased payments could be made to farmers for management of these areas.
- Ultimately, how taxes are set is the result of political negotiations and compromises, for which the results in this thesis can provide valuable input.

7 Future research

To build upon the work in this thesis, future research could focus on the missing aspects identified, to better capture the environmental sustainability of the Swedish diet in a local context. This could include establishing more reliable inventory data on emission intensities (*e.g.* for nutrients to assess eutrophication impacts) and resource use (*e.g.* to differentiate between different plant protection products to evaluate chemical pollution). Further, to link the impacts of use, further efforts should be made to develop site-dependent impact assessment methods. Moreover, better data are needed on foods imported to the Swedish market, data which could be collected together with the industry.

As regards evaluating potential goal conflicts of taxation, future work could include more detailed data on food consumption, *e.g.* for plant-based substitutes for meat and dairy, as well as different fish and seafood species. Future research could also evaluate the effects of taxation within a policy package, *e.g.* together with information-based policies. Such research should include additional aspects such as acceptance of taxation by affected stakeholders on the producer side and among consumers.

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Popular science summary

Swedish food consumption has major environmental impacts within and outside Sweden. To reduce these environmental impacts, changes in diets are needed and a climate tax on food could bring about such changes. This thesis assessed the environmental pressures of foods and produced values for application in food taxation and sustainability assessments of diets.

A critical part of the work was to determine how values of the climate impact of foods can be established for application in taxation. A consistent and transparent method was developed for calculating the climate impact of foods. Using this method, which is based on Life Cycle Assessment, overall greenhouse gas emissions from production of foods available on the Swedish market were calculated. The Life Cycle Assessment methodology involves a number of unavoidable methodological choices, some of which were evaluated in the thesis to determine the effects on climate impacts and resulting tax levels for foods. The results showed that methodological choices in LCA often involve a trade-off between achieving simplicity in calculations and establishing data resulting in a cost-efficient tax that can lead to emission reductions at the lowest cost.

Next, the global variables used in the EAT-*Lancet* report were compared against the Swedish Environmental Objectives. This comparison revealed that the EAT-*Lancet* report covers many aspects that are included in the Swedish Environmental Objectives. However, the focus of the EAT-*Lancet* variables is on the global level and indicators at finer resolution are needed to assess sustainability aspects of the diet on local level in Sweden. For example, the EAT-*Lancet* report analyses eutrophication using indicators measuring nutrient application from fertiliser application, which can cause eutrophication of neighbouring waters if nutrients leach to soil and waterways. These indicators can indicate a risk of eutrophication, but do not consider whether and to what extent nutrient application in agriculture causes eutrophication. For example, the eutrophication impacts can depend on the nutrient status of the waterway to which the nutrients are added. To better account for such aspects, methods that enable evaluation of environmental impacts on local level can be used. However, it is difficult to perform such evaluations for Swedish food consumption, as there are data gaps in trade statistics as regards impacts for imported foods. Several aspects included in the Swedish Environmental Objectives framework, such as pasture use, air pollution, acidification and chemical pesticide use, were found not to be covered by the EAT-*Lancet* variables. There are currently large data gaps in statistics on *e.g.* pesticide use which ideally should be rectified.

The 'Planetary Boundaries' for the food system show the maximum acceptable emissions and resource use for production of foods globally. In this thesis, the global boundaries were downscaled to per-capita level for the world population. The environmental pressures from the food consumed by the average Swede were then benchmarked relative to the boundaries. The environmental pressures were found to exceed global environmental targets for greenhouse gas emissions, cropland use and application of nutrients by two- to four-fold. For biodiversity impacts, the boundary was exceeded by six-fold. The only environmental category where the global target was not transgressed was freshwater use, where the pressure of the diet was well below the limit.

Meat and dairy products were found to make large contributions to several environmental aspects, such as greenhouse gas emissions and cropland use. For aspects such as pesticide use and biodiversity impacts, large shares of the overall pressures were linked to consumption of coffee and cocoa, due to their high impact per kg. Other products with a high biodiversity impact and pesticide use per kg included olive oil. Especially high biodiversity impact was seen for sheep meat from production in New Zealand. This was explained by high land use together with high biodiversity loss from occupation of land for sheep production. In comparison, beef was found to have low biodiversity impacts per kg, despite high land use, because most livestock production for the Swedish market takes place on relatively biodiversity-poor land in Sweden and Northern Europe. However, the impacts on biodiversity loss would change considerably if production were to take place in countries where the occupation of land causes higher biodiversity loss, such as Brazil.

The analyses in this thesis showed that the environmental pressures of food consumption could be reduced through climate taxation, an effect mainly explained by an overall decrease in food consumption. Due to a relatively large decline in beef consumption following taxation, pasture use was found to decrease. A decline in pasture area can be considered positive or negative, depending on the production region. In a global perspective, further expansion of agricultural land needs to be limited, as deforestation of tropical forest to clear land for grazing animals and production of *e.g.* soybean and oil palm crops causes large emissions of greenhouse gases and has severe impacts on biodiversity. From a Swedish perspective, however, grazing animals are essential for maintenance of biodiversity-rich semi-natural pastures. Increased payments could be made to Swedish farmers for maintenance of such areas, to avoid a potential goal conflict.

Ultimately, how taxes are set is the result of political negotiations and compromises. When considering climate taxation of foods, the results in this thesis can provide valuable input.

Populärvetenskaplig sammanfattning

Svensk livsmedelskonsumtion orsakar stor miljöpåverkan, både i Sverige och utomlands. För att minska miljöpåverkan behövs en förändring av våra kostmönster. Sådana förändringar skulle kunna påskyndas av en klimatskatt på mat. Arbetet i avhandlingen inkluderade att beräkna livsmedels miljöavtryck för tillämpning i beskattning av mat och för bedömning av kosters hållbarhet.

En central del i avhandlingens arbete var att undersöka hur livsmedels klimatavtryck kan beräknas för att tillämpas i beskattning. I arbetet utvecklades en konsekvent och transparent metod för att beräkna livsmedels klimatavtryck. Den metod som utvecklades för beräkningarna baseras på livscykelanalys där de sammanlagda utsläppen för produktionen av olika livsmedel som finns på den svenska marknaden togs fram. Beräkningarna kartlade de utsläpp av växthusgaser som uppstår i olika produktionssteg från åkern till butiken. Vid beräkningar med livscykelanalys behöver alltid ett antal metodval göras. I avhandlingen undersöktes hur valen påverkar livsmedels klimatavtryck och skattenivåer. Resultaten från avhandlingen visar att metodvalen ofta är en avvägning mellan att uppnå enkelhet i beräkningarna, och att ta fram data som resulterar i en kostnadseffektiv skatt som kan leda till utsläppsminskningar till lägsta kostnad.

I avhandlingen jämfördes hur de globala variabler som används i den så kallade EAT-*Lancet*-rapporten förhåller sig till det svenska miljömålssystemet. Jämförelsen visade att EAT-*Lancet*-rapporten täcker in många aspekter som ingår i det svenska miljömålssystemet. Däremot är fokus på global nivå och mer finmaskiga indikatorer skulle behövas för att täcka in hållbarhetsaspekter av kosten på en lokal nivå i Sverige. Ett exempel är övergödning där EAT-*Lancet*-rapporten använder en indikator som analyserar tillförsel av näringsämnen genom gödsling. Näringsämnen kan vid gödsling riskera att läcka ut till närliggande marker och vattendrag och orsaka övergödning. De indikatorer som används i EAT-*Lancet*-rapporten kan å ena sidan ge en indikation för övergödning men de tar däremot inte hänsyn till om, eller i vilken omfattning tillförseln faktiskt orsakar övergödning. Omfattningen av övergödningen kan t.ex. påverkas av näringsstatus på kustvatten dit näringstillförseln sker. För att bättre ta med sådana platsberoende aspekter behöver metoder användas som kan kartlägga miljöpåverkan på lokal nivå. Sådana utvärderingar är dock svåra att göra då det finns brister i statistiken för import till Sverige för att kunna kartlägga produkters ursprungsländer. Aspekter som idag ingår i det svenska miljömålssystemet men som inte täcks in av EAT-*Lancets* variabler är t.ex. betesmarksanvändning, luftföroreningar, försurning och användning av bekämpningsmedel. Det finns i dagsläget stora luckor för statistiken som ligger till grund för beräkningarna av exempelvis bekämpningsmedelsanvändning.

De ’planetära gränser’ som har uppskattats för livsmedelssystemet speglar det maximala taket globalt för utsläpp och resursanvändning för produktionen av livsmedel. I avhandlingen skalades de globala gränserna ned till per-capita-nivå för världens befolkning. Sedan undersöktes hur miljöpåverkan från den svenska konsumtionen av mat per person och år förhåller sig till dem. För växthusgasutsläpp, användning av åkermark samt kväve- och fosfortillförsel visade sig miljöpåverkan ligga två till fyra gånger över de tillåtna gränserna. Särskilt utsatt visade sig dock påverkan på den biologiska mångfalden vara där medelsvenskens nuvarande kostmönster leder till att gränsen överskrids med det sexdubbla. Den totala vattenanvändningen från kosten håller sig däremot under den uppsatta gränsen.

Konsumtionen av kött och mejeriprodukter hade störst utslag på flera aspekter såsom totala växthusgasutsläpp och markanvändning. Sett till andra aspekter som användning av bekämpningsmedel och påverkan på den biologiska mångfalden visade det sig däremot att en mängd växtbaserade produkter som kaffe och kakao orsakar höga miljöavtryck. Även olivolja visade sig ha hög användning av bekämpningsmedel och stor inverkan på den biologiska mångfalden per kg. Störst påverkan på den biologiska mångfalden hade lammkött som konsumeras i Sverige och som är importerat från Nya Zeeland. Den produktionen tar både mycket jordbruksmark i anspråk, och sker på en plats med hög artrikedom där det finns en hög risk för att arter utrotas. I jämförelse visade det sig att det nötkött som äts i Sverige, trots sin höga användning av jordbruksmark, leder till betydligt lägre påverkan på den biologiska mångfalden. Detta beror på att majoriteten av produktionen sker i områden i Sverige och norra Europa med relativt låg artrikedom. Hade produktionen däremot skett i t.ex. Brasilien hade utslaget blivit betydligt högre.

Avhandlingens resultat visar att en klimatskatt på livsmedel har potential att minska miljöbelastningen inom flera miljöområden. Minskningarna är framförallt en effekt av en minskad konsumtion av mat. På grund av minskningen i konsumtionen av nötkött minskar även användningen av betesmark. Beroende på vilka typer av marker och i vilka länder betet sker kan det ha både positiva och negativa konsekvenser. Globalt sett behöver expansionen av jordbruksmark begränsas. Skövling av tropisk regnskog för att göra plats för betesmark och för odling av grödor som soja och oljepalm leder till stora växthusgasutsläpp och påverkan på den biologiska mångfalden. I Sverige behövs en viss mängd betesdjur för att underhålla naturbetesmarker som är rika på biologisk mångfald. För att undvika en potentiell målkonflikt där en klimatskatt slår mot bete av naturbetesmark i Sverige skulle ersättningarna till naturbetesmarkerna kunna höjas vid införandet av en klimatskatt.

Hur skatter sätts är i slutändan ett politiskt beslut, baserat på diskussioner och kompromisser. Resultatet från avhandlingen kan ge värdefull input till sådana sammanhang.

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Design of a climate tax on food consumption: Examples of tomatoes and beef in Sweden

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ABSTRACT

This study examined appropriate design of efficient climate taxes on consumption of food by constructing a simple theoretical model and exemplifying the results using the examples of tomatoes and beef in Sweden. The theoretical results showed that, for the tax to be efficient, *i)* existing taxes on greenhouse gases (GHGs) should be considered when calculating the climate impact in order to avoid double taxation, and *ii)* taxes should be differentiated between GHG (here carbon dioxide (CO_2) methane (CH_4) and nitrous oxide (N_2O)) because of their differing climate impacts. The calculations of climate taxes on tomatoes and beef in Sweden indicated considerable differences in the tax level depending on whether conditions (*i*) and (*ii*) were considered or not. The commonly applied approach in the literature on climate taxes on food, i.e. taxing carbon dioxide equivalents (CO_2e) using global warming potential over 100 years (GWP_{100}) and neglecting existing taxes on GHG emissions, results in a tax that is 1.4–2.8 times higher than the efficient tax for tomatoes and a tax that is between 30% lower and 20% higher than the efficient tax for beef. Despite the relatively low variations in the climate tax on beef, estimated reduction in emissions from beef ranged between 23% and 35% depending on choice of tax. The price increases on food due to a climate tax and associated effects on emissions can thus show large variation depending on the tax calculation method.

1. Introduction

The food producing sector accounts for 19–29% of total global emissions of greenhouse gases (GHGs) (Vermeulen et al., 2012). Hence, emissions reductions are needed also in the food sector in order to meet the agreement adopted at the Paris climate conference in December 2015 to limit global warming to well below 2 °C above pre-industrial levels (UN, 2015). Emissions from agriculture include release of livestock related emission from enteric fermentation and manure management, which accounts for two third of total emissions from agriculture (FAO, 2014). In addition, processing, transport and packaging of food increase total emission levels from the sector. As potentials to reduce emissions from the production of food are limited, the importance of dietary shift for reaching climate change targets has been highlighted in many studies (e.g. by Bajželj et al., 2014; Hedenus et al., 2013; Röös et al., 2017). As changing eating patterns is challenging, policy options based on information will likely have to be complemented by economic incentive options (Garnett et al., 2015).

According to economic theory, efficient policies are defined as the policies which maximise net benefits from reductions in GHG. Conditions for an efficient tax on GHG emissions are that the tax shall; i) be the same for all sources per unit of emission and ii) correspond to the climate damage of a marginal increase in emission (e.g. Baumol and Oates, 1988). It is well known that damage from climate change is a global issue, so a globally efficient policy should include all emissions sources in the world (e.g. Baumol and Oates, 1988). Although positive steps have been taken in international negotiations on GHG reductions, a global tax is probably not likely in the near future. Countries could strive for efficient climate policy design within their territories, but may face conflicts with other high-priority goals such as competitive production. This is the case for GHG emissions from the food system, which are generated in agriculture and fisheries and in post-farm energy use. A national tax on emissions from the production of products in these sectors might lead to higher consumer prices and a shift in consumption towards imported food. If emissions from domestic production are

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relatively low compared with those from production in other countries, there is a risk of increases in overall emissions, which would offset the reduction in domestic emissions (Van Doorslaer et al., 2015). Such emissions offsetting is the main argument in favour of taxing consumption instead of production of food. Taxes on food consumption reduce emissions from both imported and domestically produced goods, thus reducing the risk of overall emissions increases.

If a climate tax is to be introduced on food consumption, the question examined in this study is how a tax on food products should be designed to meet the two conditions for efficient emissions reduction within a country. The principle of taxing a product with negative externalities is straightforward; the product should be taxed according to its emissions of GHG and the damage cost per unit emissions common to all emission sources (e.g. Baumol and Oates, 1988). Although simple in theory, calculation of both these components is more complicated in practice.

When assessing the climate impact of different food products, a number of aspects have to be considered. Food production involves a long chain of activities, with emissions arising in pre-farm activities such as the production of farm inputs, e.g. fertilisers and pesticides, on-farm activities such as rearing of animals and cultivation of food and feed crops and post-farm processes (see Clune et al. (2017) for a review). All steps in the food chain generate GHG emissions, but some of these emissions may already be included in existing carbon taxation or trading schemes, a point that needs to be considered when establishing an efficient climate tax on food in order to avoid double taxation. In addition, unlike the transport and energy sector, in which emissions of carbon dioxide (CO_2) dominate, the food system generates substantial amounts of non-taxed methane (CH_4) and nitrous oxide (N_2O) emissions (Vermeulen et al., 2012), which have different characteristics in terms of global warming than CO_2 (Myhre et al., 2013).

With respect to condition (ii) for efficient policy design, different taxes on the three GHGs will be required if the costs of their marginal climate damage differ. It is well known that their climate effects differ, with global warming potential over 100 years (GWP_{100}) (Myhre et al., 2013) being the most common way of weighting the warming effect of these gases into one common unit, CO_2-equivalents (CO_2e). However, the use of GWP_{100} has been questioned and other metrics have been suggested (Persson et al., 2015). In addition, studies show that the damage costs of marginal emissions of the three GHGs differ (e.g. Marten et al., 2015).

A number of studies have examined the impact of climate taxes on consumption of food in terms of consumer responses and associated effects on GHG emissions from a price increase through a consumption tax on food (Wirsenius et al., 2011; Edjabou and Smed, 2013; Säll and Gren, 2015; Abadie et al., 2016; Caillavet et al., 2016; Chalmers et al., 2016; Springmann et al., 2017). Apart from the works by Abadie et al. (2016) and Caillavet et al. (2016), the suggested tax in all studies is based on data calculated using life cycle assessment (LCA) methodology, without considering existing taxes in the production chain, and all studies use GWP_{100} to weight the different GHGs (CO_2, CH_4, N_2O) into CO_2e.

In this study, we examined the conditions in which these two practices used in the literature fulfil the conditions for efficient climate taxes on food products, and the implications of deviations. To this end, we constructed a theoretical model to derive conditions for an efficient climate tax on food products (Section 2). We then applied the theoretical results in calculation of efficient consumption taxes on tomatoes and beef sold in Sweden and compared these with non-efficient GHG taxes (Sections 3 and 4). These two food items were chosen as they have different forms of production chain, existing GHG taxes and type of GHG generated. The impacts of different calculation methods on the emission reductions caused by the tax are examined in Section 5. We discuss our results and present some conclusions in section 6.

2. Simple theoretical analysis of efficient climate taxes on food products

Almost a century ago, Pigou (1920) suggested that an efficient environmental tax on a product should reflect its marginal environmental damage. For a climate tax, this means that each product should be taxed according to its emissions of GHGs and the marginal damage cost of the GHG. The implication of this condition is that taxes should be differentiated over time, since the climate damage of a particular emission changes over time because of the decay rate in the atmosphere and the discount rate, which makes future abatement less costly than current (e.g. Nordhaus, 2007; Tol, 2013). For the purposes of demonstration without loss of generality, in this study we considered only the design of a tax on food products in a particular time and constructed a static model.

The use of LCA data as a basis for taxes without considering existing taxes on emissions in different stages of the life cycle results in double taxation of those emissions. This violates the first condition for an efficient tax, i.e. that the tax per unit emissions should be the same for all sources. Furthermore, if the marginal climate impact differs between different types of GHGs, the associated emission taxes should also differ to reflect this. All previous studies on consumption taxes on food disregard existing taxes in the production chain and use GWP_{100} for weighting GHG (Wirsenius et al., 2011; Edjabou and Smed, 2013; Säll and Gren, 2015; Abadie et al., 2016; Caillavet et al., 2016; Chalmers et al., 2016; Springmann et al., 2017). This approach gives an efficient tax on food only if the different stages in the production cycle are not subject to any climate taxes and only if a single type of GHG is generated in the system.

In order to illustrate this, we constructed a simple model of a profit-maximising firm, which is a common assumption of a firm's behaviour in economics. We assume competitive markets where the output and input prices are given to each firm. For the firm, incomes are obtained from sales of food, Q at price p. Q is a function of all inputs needed to produce the food, I^F where F = 1...N inputs, written as $Q = Q(I^1 ... I^N)$. The inputs can be labour, capital and intermediate products such as electricity and transport. The unit price of the inputs is k^F. In addition to costs of inputs, the firm pays taxes, tax^G, on emission of GHG gases, P^G, where G = 1...H are the different GHG gases. Here we assigned specific GHG taxes, tax^G, determined by the cost of their marginal impact on warming of the atmosphere (e.g. Nordhaus, 2007). For the firm to maximise its profit, π, by choosing I^F the following applies:

$$\underset{I^1,..,I^N}{Max} \quad \pi = pQ - \sum_F k^F I^F - \sum_G tax^G P^G(Q) \tag{1}$$

The first-order conditions are:

$$\frac{\partial \pi}{\partial I^F} = p\frac{\partial Q}{\partial I^F} - k^F - \sum_G tax^G \frac{\partial P^G}{\partial Q}\frac{\partial Q}{\partial I^F} = 0 \tag{2}$$

The firm chooses I^F where the value of the marginal product equals the unit cost of the input plus tax payments from the marginal increase in emissions. The crucial issue is now how the price of the inputs, k^F, is determined. For demonstration purposes without loss of generality, we disregarded the cost associated with the deliveries to the suppliers of inputs to the firm and assumed that a subcontractor's costs consist of production costs, C^F, and climate tax payments. The associated profits of a subcontractor, π^F, are then determined by maximisation of:

$$\pi^F = k^F I^F - C^F\left(I^F\right) - \sum_G tax^G P^{G,F}\left(I^F\right) \tag{3}$$

The profit-maximising subcontractor will then choose a quantity I^F where:

$$k^F = \frac{\partial C^F}{\partial I^F} + \sum_G tax^G \frac{\partial P^{G,F}}{\partial I^F} \tag{4}$$

Inserting this expression for k^F into (2) we obtain:

$$\frac{\partial \pi}{\partial I^F} = p\frac{\partial Q}{\partial I^F} - \left(\frac{\partial C^F}{\partial I^F} + \sum_G tax^G \frac{\partial P^{G,F}}{\partial I^F}\right) - \sum_G tax^G \frac{\partial P^G}{\partial Q}\frac{\partial Q}{\partial I^F} = 0 \tag{5}$$

Expression (5) shows that when climate taxes are imposed on all firms in the production chain, each firm pays according to its own emissions. However, it is common practice in the literature on climate taxes on food products to assign total GHG emissions from the subcontractor to the food-producing firm, which gives:

$$\frac{\partial \pi}{\partial I^F} = p\frac{\partial Q}{\partial I^F} - k^F - \sum_G tax^G \left(\frac{\partial P^{G,F}}{\partial I^F} + \frac{\partial P^G}{\partial Q}\frac{\partial Q}{\partial I^F}\right) = 0 \tag{6}$$

The food-producing firm thus pays for GHG emissions which are already paid for by the subcontractors. There is no double counting of climate tax payments only if none of the suppliers pays climate tax for any GHG, since then $k^F = \frac{\partial C^F}{\partial I^F}$ for all suppliers.

When all firms pay climate taxes, it is generally assumed in the literature that they pay the same tax. If this is not the case, adjustments need to be made. For example, if the taxes paid by the suppliers, $tax^{G,F}$ are lower than those paid by the food-producing firm, tax^G, the corresponding adjustment would be $\sum_G (tax^G - tax^{G,F})\frac{\partial P^{G,F}}{\partial I^F}$. That is, the climate taxes are increased on the food product corresponding to the differences in climate tax payments and GHG emissions caused by the inputs from each supplier. Admittedly, the associated increase in the tax at the supplier level might be difficult to implement in practice, but needs to be computed for efficient taxation. On the other hand, if the supplier pays higher climate unit taxes than the food-producing firm, tax payments need to be refunded. In principle, this system is not much different from the excise duty or VAT system where a seller pays all VAT on the sales and deducts the VAT payments made for the purchases of inputs. The difference is that in many countries the VAT is a constant percentage of the total sales, whereas the GHG tax in our system is constant per unit GHG emission.

So far, we have not considered imports of the product in question. The argument in favour of a tax on consumption necessitates the same climate tax on imports and domestically produced products. In principle, the output from producers in each country should then be taxed as shown by equations (1)–(5). If existing taxes in the exporting countries deviate from those in the importing country and there is a desire to assign the same tax on GHG emissions in the importing country, the taxes on imported food items must be adjusted according to the difference in tax levels between the countries. The country then assigns the same tax on emissions from food consumed in the country, irrespective of origin of production. Similarly to differences in tax payments between suppliers and the food-producing firm, for the tax level to be efficient according to the first principle, a mechanism that taxes or refunds differences in tax payments between the imported and domestic food products needs to be implemented.

With respect to the second condition of efficient climate tax, a practice used by all previous studies examining effects on consumption of a tax on food products is to use GWP_{100} to weight the different GHG, instead of assigning separate taxes for each GHG (Wirsenius et al., 2011; Edjabou and Smed, 2013; Säll and Gren, 2015; Abadie et al., 2016; Caillavet et al., 2016; Chalmers et al., 2016; Springmann et al., 2017). Whether or not such a tax on merged GHG generates the same total tax on food products as differentiated GHG taxes depends on the relationship between the GWP_{100} weights and tax^G, which reflects the cost of marginal damage. Since CO_2 is used as the denominator, the CO_2e tax for a specific GHG, $tax^{CO_2e,G}$, is calculated as $tax^{CO_2e,G} = tax^{CO_2}\omega^G$, where ω^G is the weight of emission G in relation to CO_2, for which $\omega^{CO_2} = 1$. Using GWP_{100} (Myhre et al., 2013), the weight for CH_4 is 34 (ω^{CH_4}=34) and that for N_2O is 298 (ω^{N2O}=298). The $tax^{CO_2e,G}$ will then be the same as tax^G only when $\omega^G = \frac{tax^G}{tax^{CO_2}}$ for all $G = 1,..,H$, i.e. when the marginal damage in monetary terms of GHG G in relation to CO_2 is the same as ω^G.

3. Calculation of GHG emissions from production of tomatoes and beef consumed in Sweden

In order to illustrate how the different climate tax design options affect tax levels, we calculated taxes for two different food products, tomatoes and beef. These products differ with respect to sources of GHG, where tomatoes mainly give rise to CO_2 emissions and beef mainly to CH_4 emissions. These differences mean that the food products differ with respect to use of existing taxes on emissions from inputs and in terms of the implications of separate GHG taxes compared with a tax on CO_2e.

We calculated cradle-to-retail carbon footprints for the production of 1 kg of tomatoes and 1 kg bone-free beef meat. We used an attributional approach (Finnveden et al., 2009), as it is argued to be a better approach in policy implementation than policy evaluation (Brandão et al., 2014). Emissions from each GHG were accounted for, as were the weighted CO_2e values based on GWP_{100} from the latest IPCC report, i.e. a characterization factor for CH_4 of 34 (ω^{CH_4}=34) and for N_2O of 298 (ω^{N2O}=298) (Myhre et al., 2013). We followed the sequence of analysis in Section 2 and calculated emissions of CO_2, CH_4 and N_2O from the different inputs, distinguishing between taxed, partly taxed and non-taxed emissions. As 86% of tomatoes and 47% of beef are imported to Sweden, we performed the calculations for domestic production and for the major import countries. We applied a top-down approach in which we used official national statistics for the most influential parameters and set standard values for parameters with only a minor influence on the results, to calculate country-specific average climate impacts per kg of product. For a detailed description of the calculations, see Moberg et al. (forthcoming).

The GHG emissions from international transport were based on the distance from the capital of the country of origin to the capital of Sweden. Transport within the exporting countries was excluded due to the difficulty in obtaining data on transport from farms scattered around the producing countries to processing sites and retailers. For simplicity, we also excluded releases or sequestration of CO_2 from soil and land use change.

3.1. GHG emissions from tomato production

In Sweden, 14% of tomatoes consumed are produced in the country, 56% are imported from the Netherlands, 18% from Spain and 12% from other countries (five-year average data from Statistics Sweden, 2016). Production of tomatoes in Sweden and the Netherlands mainly takes place in heated greenhouses (SBA, 2015; van der Velden and Smit, 2016), while unheated greenhouses are commonly used in Spain (Montero et al., 2017). GHG emissions arise from the production of fertilisers and pesticides and from soil during cultivation, electricity and fuel used for heating, lighting, constructing and maintaining greenhouses and other capital goods, and packaging and transport of the products. Data and data sources are presented in Tables A1 and A2.

With respect to GHG taxes, all three tomato-producing countries are members of the European Union (EU), which means that several production sectors are part of the EU emissions trading system (ETS) and subject to associated pricing of CO_2 emissions. This affects GHG emissions from the production of fertilisers and other inputs bought from sectors included in the EU ETS. In addition, the producing countries impose national taxes on CO_2 emissions from non-ETS sectors, in particular on transport. We therefore distinguished between GHG emissions taxed by national and ETS systems. Emissions from heating is the main source of emissions from tomato production in Sweden and the Netherlands, and is also subject to national taxation in both countries. However, both countries have a complex system for taxation of different users of energy. Since the agricultural sector is not included in the EU ETS, we therefore assumed that emissions from heating are subject to the national CO_2 tax. Results of the calculations are shown in Table 1 for three different categories: 1) non-taxed, 2) taxed by a national tax and 3) covered by the EU ETS, and total emissions per kg tomatoes.

As Table 1 shows, there are considerable differences in GHG emissions between the countries, with tomatoes from the Netherlands generating highest emissions of all GHGs. Whether or not this also results in a higher climate tax per kg/tomatoes depends on the differences in the chosen reference tax level and the national and ETS prices of carbon, as discussed in Section 4.

3.2. GHG emissions from beef production

The share of domestically produced beef corresponds to approximately 53% of Swedish consumption (SBA, 2016b). Beef is imported mainly from Ireland (28% of imports), but also from the Netherlands, Germany and Poland (SBA, 2016b). Beef is produced in diverse production systems, where a common system in both Sweden and Ireland is suckler production (Hallström et al., 2014), and we therefore illustrate taxes on beef from this system produced in Sweden and Ireland. Emissions arise from manure handling, feed production, enteric fermentation from the digestive system of the ruminants, electricity and fuel use for machinery and housing, and packaging and transport of the products. Data and data sources are presented in Tables A3.

Similarly to tomatoes, we distinguished between non-taxed and taxed (national and EU ETS) GHG emissions from suckler production in the two countries (Table 2).

Unlike GHG emissions from tomatoes produced in different countries, all GHG emissions categories are relatively similar for beef produced in Ireland and Sweden. The untaxed part of total emissions is relatively high in both countries because of emissions of CH_4 from ruminant digestion.

Table 1
Greenhouse gas (GHG) emissions from tomatoes on the Swedish market (g GHG/kg tomato).

GHG and country of origin	Non-taxed	Taxed: National	EU ETS	Total
CO_2				
Sweden	0.00	632[a]	90.7[b]	723
Netherlands	0.00	1308[c]	86.6[b]	1395
Spain	0.00	287[d]	91.7[b]	379
CH_4				
Sweden	1.46			1.46
Netherlands	5.32			5.32
Spain	0.23			0.23
N_2O				
Sweden	0.09		0.04[e]	0.13
Netherlands	0.08		0.03[e]	0.11
Spain	0.08		0.05[e]	0.13
CO_2e				
Sweden	89.5	632	90.7	812
Netherlands	213	1308	86.6	1608
Spain	46.5	287	91.7	426

[a] Emissions from energy use and transport, of which 56% receive a tax deduction of 20%.
[b] Emissions from production of inputs (fertiliser, pesticides, capital goods) and packaging.
[c] Emissions from energy use and transport, carbon tax with no exemptions for the transport sector (Vollebergh, 2015).
[d] Emissions from transport, carbon tax on vehicles related to transport load (Zane, 2013).
[e] Emissions of N_2O from mineral fertiliser production.

4. Climate taxes on tomato and beef consumption

Emissions of GHG caused by the products was one component in determination of a tax per unit food item. We also had to choose a tax per unit GHG. There is a large body of literature on estimation of a carbon tax. As suggested by Gren et al. (2017), these studies can be classified into four main categories where estimates are based on: damage from GHG, balancing of costs and benefits of GHG, cost-effectiveness analysis of reaching climate targets, and revealed preferences in terms of actual carbon pricing as taxes or equilibrium prices on emissions trading markets. There is a wide range in published estimates within and between categories, i.e. 6 to 276 USD/ton CO_2e in 2015 prices, mainly depending on choice of discount rate and timing of taxes (Gren et al., 2017). A lower discount rate means higher current and future taxes, and optimal taxes usually rise over time until the emissions reduction target or equilibrium level of emissions is reached.

In our calculations, we used the actual tax on CO_2 in Sweden on all consumption, since this tax would most likely be imposed on CO_2 emissions from food consumption. This tax, which gives $tax^{CO_2} = 129$ USD/ton CO_2, was introduced in the early 1990s and is among the highest actual carbon taxes in the world. However, the agriculture sector in Sweden currently receives a 20% deduction in this tax, which we corrected for. Taxes on imported food were adjusted according to the difference in the Swedish tax and the tax in the exporting country, as discussed in Section 2. According to Vollebergh (2015), the CO_2 taxes in the Netherlands and Spain amount to 82 and 42 USD/ton CO_2, respectively. The corresponding tax in Ireland is 21 USD/ton CO_2 (Carbon Tax Center, 2017). A feature shared by all producing countries is that some of the inputs, fuel

Table 2
Greenhouse gas (GHG) emissions from production of beef (bone-free) (kg GHG/kg beef).

GHG and country of origin	Non-taxed	Taxed: National	EU ETS	Total
CO_2				
Sweden	0	3.05[a]	1.74[b]	4.79
Ireland	0	3.81[c]	3.47[b]	7.29
CH_4				
Sweden	0.92			0.92
Ireland	1.13			1.13
N_2O				
Sweden	0.02		0.01[d]	0.03
Ireland	0.03		0.01[d]	0.04
CO_2e				
Sweden	38.5	1.46	4.58	44.6
Ireland	47.4	3.99	4.80	56.2

[a] Emissions from energy use and transport, with a 20% tax deduction.
[b] Emissions from fertiliser and pesticide production, packaging and material inputs.
[c] Carbon Tax Center (2017).
[d] Emissions of N_2O from mineral fertiliser production.

and electricity, are bought from sectors included in the EU ETS system. The EU ETS allowance price fluctuated between 5 and 32 USD/ton CO_2 during the period 2007–2015 (Sandbag, 2017). In this study we applied the average, 18.5 USD/ton CO_2.

To the best of our knowledge, there are no taxes on CH_4 and N_2O in any of the producing countries, although emissions of N_2O from fertiliser production are covered by EU ETS. Two studies have calculated the marginal social cost of these emissions using integrated assessment models (Waldhoff et al., 2011; Marten et al., 2015). Their estimate for CH_4 varies between 310 and 7500 USD/ton CH_4 and that for N_2O between 5260 and 87,300 USD/ton N_2O, depending on model choice and discount rate. Their estimate for the marginal social cost of CO_2 (range 7–112 USD/ton) is lower than our chosen cost (the Swedish CO_2 tax of 129 USD/ton) and their estimated cost of climate damage from the other GHG differs from that for CO_2. Here we used the damage cost of CH_4 and N_2O in relation to CO_2 (discount rate of 2.5%) to calculate the tax on CH_4 and N_2O emissions. Using the relationship between damage cost of CH_4 and CO_2, and between the damage cost of N_2O and CO_2 in Marten et al. (2015), we obtain $tax^{CH_4} = 4123$ USD/ton CH_4 and $tax^{N_2O} = 50{,}536$ USD/ton N_2O in 2015 prices. Applying the corresponding relationships in Waldhoff et al. (2014) gives $tax^{CH_4} = 5768$ USD/ton CH_4 and $tax^{N_2O} = 79{,}500$ USD/ton N_2O. The calculated efficient taxes, i.e. the reference case, for tomatoes and beef from different countries are presented in Table 3.

For both tomatoes and beef, the tax was lowest on products produced in Sweden. The reason for the relatively low tax on tomatoes is the deduction of taxed inputs and greater use of biofuels for heating, while the national tax was lower for tomatoes produced in the Netherlands. The low tax for tomatoes produced in Spain is due to the cultivation without heated greenhouses, thus not producing emissions from heat generation (Table 1). The climate tax on Swedish beef was slightly lower (7%) than that of imports from Ireland when data from Marten et al. (2015) were used and 18% lower when data from Waldhoff et al. (2014) were used. The difference is attributable to relatively higher CH_4 and N_2O taxes in relation to the CO_2 tax in Waldhoff et al. (2014).

For both these measurements of climate damage, the relations $\frac{tax^{CH_4}}{tax^{CO_2}}$ and $\frac{tax^{N_2O}}{tax^{CO_2}}$ clearly deviated from the GWP_{100} weights of 34 and 298, respectively. The weights were 32 and 392, respectively, for the Marten et al. (2015) data and 45 and 616, respectively, for the Waldhoff et al. (2014) data. The estimated taxes on tomatoes and beef then differed for separate taxes on GHG compared with a tax applied to the CO_2e. Another source of inefficient taxation is the lack of correction for existing taxes on GHG, with the associated risk of double taxation. In order to illustrate the implications on the tax level of these two sources of inefficiency, we calculated taxes under three inefficiency scenarios;

Table 3
Tax levels for tomatoes and beef in the reference case with correction for taxed inputs and differentiation of taxes among GHG with two alternative weights of marginal climate damage, USD/ton product.

	Marten et al. (2015) [a]	Waldhoff et al. (2014) [b]
Tomatoes		
Sweden	37	42
Netherlands	96	107
Spain	40	43
Beef		
Sweden	5442	7567
Ireland	5877	9289

$tax^{CO_2} = 129$ USD/ton CO_2.
[a] $tax^{CH_4} = 4123$ USD/ton CH4 and $tax^{N_2O} = 50{,}536$ USD/ton N2O (Marten et al., 2015).
[b] $tax^{CH_4} = 5768$ USD/ton CH4, $tax^{N_2O} = 79{,}500$ USD/ton N2O (Waldhoff et al., 2014).

1) Differentiation of GHG taxes, but no correction for taxed inputs
2) No differentiation of GHGs (using GWP_{100} to weigh GHGs into CO_2e), but with correction for taxed inputs.
3) No differentiation among GHGs (using GWP_{100} to weigh GHGs into CO_2e) and no correction for taxed inputs.

When calculating the tax without differentiation of taxes between GHG (scenarios 2 and 3), we used the Swedish tax of 129 USD/ton CO_2e. The results for tomatoes are presented in Fig. 1.

In the worst case, the inefficient tax was almost three times as high as the reference tax (Fig. 1). This occurred when there was no correction for taxed inputs, irrespective of differentiation of taxes among GHG. This was of particular relevance for Sweden (Fig. 1), because of the higher existing CO_2 tax covering energy use than in the Netherlands and Spain. It is also noteworthy that in scenario 2, with correction for taxed inputs but no differentiation of GHG taxes, the tax per ton tomatoes was close to the reference tax (see Table 3). This illustrates the dominance of CO_2 emissions in relation to CH_4 and N_2O in tomato production.

The pattern changed when the level of the inefficient taxes in relation to the reference tax was plotted for beef (Fig. 2).

The highest deviation from the reference tax on beef was a reduction of almost 30% in the scenario with no differentiation of GHG and use of the Waldhoff et al. (2014) estimates of climate damage. The taxes based on CO_2e were then too low, but were of the same order of magnitude as the reference tax when data from Marten et al. (2015) were applied. There was a slight difference between meat from Ireland and Sweden (Fig. 2), because of the higher non-taxed emissions of CH_4 and N_2O per ton meat in Ireland.

5. Emission implications of different taxes on beef

While climate tax levels differ depending on calculation method, their imposition on food consumption products may have little impact if the tax is low relative to the price of the commodities in Sweden. The introduction of differentiated taxes depending on country of origin will generate changes in profit margins for the firms and reallocate their sales depending on changes in demand and consumer prices. Consumers will respond to the tax on the commodity by reducing demand on that product, but demand for other food items will also be affected, how depending on if they are complements or substitutes. The assessment of the net effect of all these adjustments in Sweden would require a partial equilibrium model of the agricultural sector, which is not available. Instead, estimates are made of consumers' demand for meat products (beef, pork, and chicken) in Sweden (Säll and Gren, 2015), which we use to illustrate impacts on demand and associated CO_2e emissions of introduction of the calculated taxes on beef in Sweden. In the following, we illustrate the impact of the different climate taxes for an introduction of the tax calculated for beef produced in Sweden and estimate associated impacts on CO_2e emissions. This is likely to underestimate the effects since emission intensity is higher for beef produced in Ireland.

The exclusion of tomatoes is motivated, not only because of the lack of data on demand, but also because of the relatively low emissions and climate tax. Total consumption of tomatoes in Sweden was approximately 10 kg in 2015 (SBA, 2015), and the calculated CO_2e emissions per kg varies between 0.4 and 1.6 depending on country of origin (Table 1) which gives a total of maximum 0.016 ton CO_2e per capita. Per capital consumption of beef amounts to approximately 18 kg bone-free meat per year (Table A5) and the emissions for the average beef meat (including both meat from suckler production (Table 2) and meat from dairy production) are approximately 36 kg per kg bone-free meat (Moberg et al., forthcoming), which gives a total of 0.65 ton CO_2e.

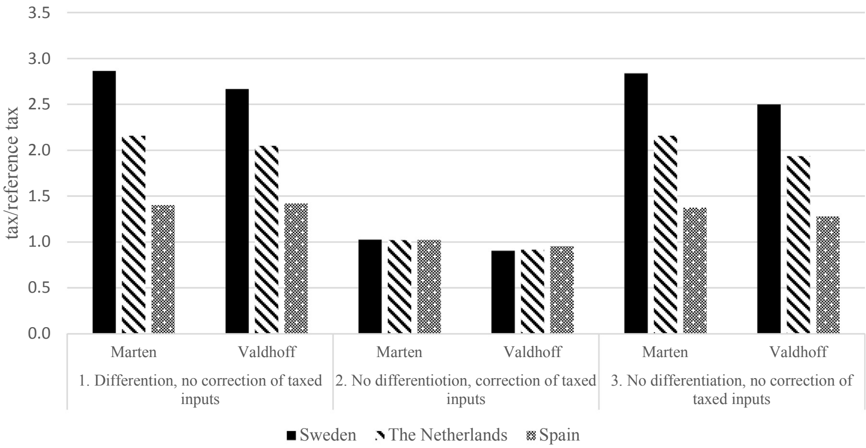

Fig. 1. Calculated climate consumption tax per ton tomatoes under three different inefficiency scenarios compared with the reference (efficient) tax for tomatoes produced in Sweden, the Netherlands and Spain and for two assessments of the climate impact of CH_4 and N_2O in relation to CO_2 (Marten et al., 2015 (Marten); Waldhoff et al., 2014 (Waldhoff)).

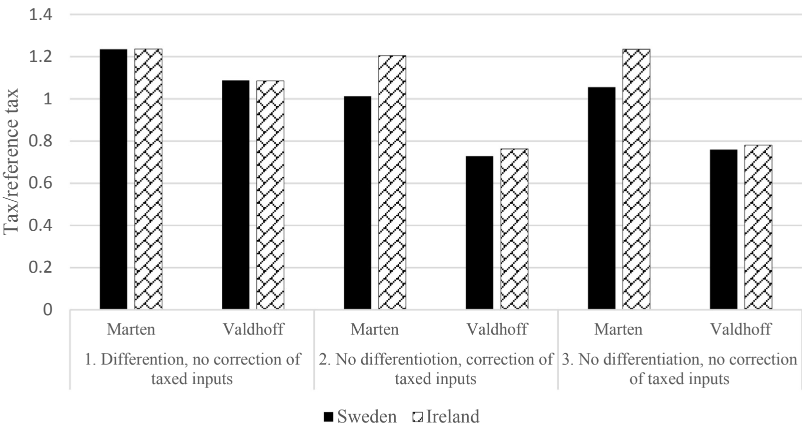

Fig. 2. Calculated climate consumption taxes per ton beef under three different inefficiency scenarios compared with the reference (efficient) tax for beef produced in Sweden and Ireland and for two assessments of the climate impact of CH_4 and N_2O in relation to CO_2 (Marten et al., 2015 (Marten); Waldhoff et al., 2014 (Waldhoff)).

Depending on the impact of the climate tax on the consumer price and associated effect on the demand, the impact on emissions can be considerable. The approximate price of beef was 18,310 USD per ton bone-free beef in 2015 (Säll and Gren, 2015) The increase in the price of beef owing to a climate tax would then range between 30% and 45%, depending on calculation method (Table 4). The impact of a climate tax on demand for beef is measured by the own price elasticity, which shows the change in demand in percent from a unit percent change in the price of beef. This elasticity amounts to −0.538 (Table A5), which implies that the lowest tax level would reduce demand by 2.91 kg beef, and the CO_2e emissions by 0.105 ton per capita. The inclusion of cross price effects, i.e. effects on demand for pork and chicken from the price increase in beef has a minor impact on the effects by raising the emission reduction to 0.106 ton CO_2e. However, when Valdhoff data are used for differentiating the taxes between the GHGs, the reduction increases to 0.147 ton (Table 4).

The calculated emission reduction varies between 0.106 and 0.160 ton/capita or between 16% and 25% of untaxed emissions from beef depending on calculation method, which reflects the range in the level of climate taxes. The main difference in impacts is due to choice of weighting of the GHGs. However, a move from the commonly used taxation in column 5 to the efficient tax in column 2 has minor impacts when we use Marten et al. (2015) data on social cost, but raises the reduction by 31% when Waldhoff et al. (2011) measures are used.

Table 4
Tax per kg, reductions in emission per capita and in % of emissions from beef of alternative calculation methods for climate taxes on bone-free beef in Sweden.

	Efficient tax; Marten Waldhoff		Differentiation of GHG but no corrections for inputs; Marten Waldhoff		No differentiation of GHG, but with input correction	No differentiation of GHG, and no input correction
Tax, USD/kg beef	5442	7567	6720	8230	5508	5743
CO_2e red. ton/capita	0.106	0.147	0.130	0.160	0.108	0.112
% of CO_2e from beef	16	23	20	25	17	17

6. Discussion and conclusions

The main purpose of this study was to establish the conditions for efficient design of a climate tax on food consumption. To this end, we constructed a simple theoretical model, which showed that the tax should rest on the existence of other climate taxes, such as carbon taxes on fossil fuels, in food-producing countries. Calculating the climate tax without considering existing taxes on GHG emissions can lead to double taxation of part of the emissions from a food-producing firm and hence non-optimal (too high) taxes. The magnitude of the deviation from the efficient tax level depends on the GHG emissions from the inputs and the tax level of these emissions. Another main theoretical conclusion was that the tax should be differentiated between the three main GHGs (CO_2, CH_4, N_2O), because of differences in marginal climate impact. It was shown that the commonly applied method of merging the different GHGs into CO_2e using the GWP_{100} methodology gives the same result as taxes differentiated among GHGs only under specific conditions on the chosen weights.

On calculating climate taxes on tomatoes and beef (from suckler production), relatively large differences were found between the efficient and non-efficient taxes on tomatoes, i.e. when existing taxes on emissions were not considered and/or a common tax on CO_2e was assumed. The inefficient tax was up to three times as high as the efficient tax, mainly owing to non-correction for existing taxes. The magnitude of inefficiency was smaller for beef, where the inefficient tax was too low, corresponding to approximately 70% of the efficient level when assuming a common CO_2e tax instead of differentiation among GHG. The impact on emissions depends on the tax level and the consumer and producer responses to the tax. It was shown with a simple illustration that the price increase of beef owing to the climate tax varies between 30% and 45% depending on the choice of tax, and the calculated emission reduction ranged between 16% and 25% in the CO_2e emissions from beef consumption. However, these impacts were calculated based on the assumption of competitive markets for beef, the impact can be smaller when there is market power where the price mechanism not fully reflects changes in costs of the good (e.g. Cararro et al., 1996).

These results can be transferred to other food items and other products subject to a consumption tax based on life cycle emissions. For given emissions levels, the magnitude of difference in taxes between calculation methods depends on the share of taxed versus non-taxed emissions, where CO_2 emissions are taxed in several countries and CH_4 and N_2O emissions are not. For food items with a high share of taxed CO_2, the deviations between efficient and inefficient taxes depend on allocation of inputs and differences in tax level between the inputs and the chosen CO_2 tax. When a relatively large part of the emissions are not taxed, which is the case for CH_4 and N_2O emissions in most countries, the deviation between an efficient and inefficient tax is explained by the relationship between marginal damage of the GHG and GWP_{100} weights.

It must be acknowledged that our analysis and calculations rest on some strong assumptions. Whether or not it is possible to implement efficient taxes in practice will depend on several factors. One is that it is must be possible to implement GHG taxes on food consumption depending on country of origin. In our example, there would be different consumption taxes on tomatoes and beef produced in Sweden and other countries. For example, the tax on tomatoes from the Netherlands could be twice that on tomatoes produced in Sweden. Such country-specific taxes might violate trade agreements (Bähr, 2015).

Another complicating factor with implementing efficient taxes on food consumption is the transaction cost that comes with calculating them for a range of different food products. Here we used tomatoes and beef to illustrate how inefficient taxes differ from efficient taxes, but foods on globalised food markets such as that in Sweden and other industrialised countries are produced from hundreds of different commodities in thousands of different food products. Tracing raw materials from different countries, keeping track of all emissions throughout the life cycle and determining which of these emissions are covered by existing taxes would require considerable effort and might be associated with high costs. Thus considering the transaction costs, it might be less costly for society to implement a simpler (inefficient) tax scheme. Insights gained in this study would be valuable in that case. For example, a decision might be made to tax only emissions known to be untaxed world-wide, i.e. CH_4 and N_2O emissions, hence avoiding double taxation of CO_2 emissions. This would give a tax close to the efficient level, although the tax would risk being too low for products imported from countries where CO_2 taxes on energy use are not in place. The most favourable taxation strategy would thus depend on the main country of origin of imports and the tax levels in these exporting countries in relation to the domestic CO_2 tax.

Using differentiated taxes for CO_2, CH_4 and N_2O based on either Marten et al. (2015) or Waldhoff et al. (2014), rather than applying the CO_2 tax to emissions aggregated into CO_2e, is straight-forward to calculate and could be implemented without high transaction costs, as it only means using different weights for CH_4 and N_2O rather than a single value. However, the diversity in estimates obtained here for two different food products is a challenge (Figs. 1 and 2). It could be argued that the GWP_{100} weights now used extensively in policy may be an option that is more acceptable to policy makers and other stakeholders, despite this leading to an inefficient tax.

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We are much indebted to two anonymous reviewers for valuable comments, and to the Swedish Environmental Protection Agency for funding of the project ‘Effects of climate tax on the food including recycling of the income’ (grant NV03211-15).

Appendix

Table A1
Data sources for calculations of greenhouse gas (GHG) emissions in the production chain for tomatoes in Sweden, the Netherlands and Spain

Source of GHG emissions	Sweden	The Netherlands	Spain
Fertiliser			
Quantity used	Davis et al. (2011)	Ecoinvent Centre (2016)	Ecoinvent Centre (2016)
Emissions factor	Yara (2010)	Yara (2010)	Yara (2010)
Pesticides			
Quantity used	Statistics Sweden (2010)	Ecoinvent Centre (2016)	Ecoinvent Centre (2016)
Emissions factor	Ecoinvent Centre (2016)	Ecoinvent Centre (2016)	Ecoinvent Centre (2016)
Heating of greenhouses			
Quantity used	SBA (2014)	van der Velden and Smit (2016)	
Emissions factor	Gode et al. (2011)	Gode et al. (2011)	
Direct and indirect[a] emissions of N_2O from soil	IPCC (2006)	IPCC (2006)	IPCC (2006)
Production of capital goods	Frischknecht et al. (2007)	Frischknecht et al. (2007)	Frischknecht et al. (2007)
Packaging	National Food Agency (2011), Nilsson et al. (2009)	National Food Agency (2011), Nilsson et al. (2009)	National Food Agency (2011), Nilsson et al. (2009)
International transport	Own calculations[b]	Own calculations[b]	Own calculations[b]

[a]Assumed to be 10% of direct emissions. In Swedish greenhouses, much of the drainage water from greenhouses is recirculated. In the Netherlands, recycling of drainage water is compulsory (Röös and Karlsson, 2013). In these closed systems, no nutrient leaching takes place and indirect emissions of N_2O were thus not accounted for.
[b]Calculations based on distances between the capital city in the countries and NTMCalc Environmental Performance Calculator by the Network for Transport and Environment (NTM, n.d.).

Table A2
Data used in calculations of the climate impact of tomatoes on the Swedish market

	% of Swedish consumption	Yield (kg/ha)	Pesticide use (kg/ha)	Fertiliser use (kg/ha)	Energy sources in greenhouse (%)	Energy use in greenhouse (GJ/ha)
Sweden	14[a]	311 189[b]	7.5[c]	1307[d]	Wood chips (43), district heating (28), natural gas (15), oil (5), other (9)[e]	9 500[e]
Netherlands	56[b]	407 243[f]	38.5[g]	864.2[g]	Natural gas (95), renewable energy sources (5), [h]	11 000[h]
Spain	18[b]	89 295[f,i *]	14.8[g]	273.8[j]	–	–

[a]SBA (2016a); [b]SBA (2015); [c]Statistics Sweden (2010); [d]Davis et al. (2011); [e]SBA (2014); [f]FAOSTAT (2017); [g]Ecoinvent Centre (2016); [h]van der Velden and Smit (2016); [i]Eurostat (2017); [j]Gobierno de España (n.d).
*Average of yields in greenhouses and open field.

Table A3
Data sources for calculations of greenhouse gas (GHG) emissions in the production chain for beef in Sweden and Ireland

Source of emissions	Sweden	Ireland
Suckler production systems (number of head, slaughter age, slaughter weight, feed consumption and grazing period)	Cederberg et al. (2009) with update on protein in feed from L. Ehde (pers. comm. 2017)	Cederberg et al. (2009) with update on protein in feed from L. Ehde (pers. comm. 2017)
Nitrogen excretion/animal	SEPA (2016)	EPA, Ireland (2016)
GHG emissions from feed	SIK/SP, n.d.	SIK/SP, n.d.
Manure management		
Quantity	Cederberg et al. (2009)	Cederberg et al. (2009)
Emissions factor	IPCC (2006)	IPCC (2006)
Electricity		
Use of fuel	Edström et al. (2005)	SEAI (2016)
Emission factor	Gode et al. (2011)	Gode et al. (2011)
Packaging	National Food Agency (2011), Nilsson et al. (2009)	National Food Agency (2011), Nilsson et al. (2009)
International transport[a]		Own calculations

[a]Calculations based on distance between capital city in the countries and NTMCalc Environmental Performance Calculator by the Network for Transport and Environment (NTM, n.d.).[b]Emissions from electricity use calculated using the Irish electricity mix.

Table A4
Data used in calculations of the climate impact of beef on the Swedish market

	Slaughter age (months)[a]	Slaughter weight (kg)[a]	Feed consumption including feed waste: roughage fodder/pasture/grain/concen-trate (kg)[a]	Grazing period (months)	N excreted in manure (kg)
Sweden	20	298	1963/970/379/250	9	106.3
Ireland	24	271	3032/783/118/24	7.5[c]	74.4

[a]Cederberg et al. (2009).

Table A5
Price elasticities, consumption/capita, and CO2e emissions from beef, pork, and chicken. sold in Sweden

	Elasticity[a]; Beef Pork Chicken			Consumption, kg bone-free/capita[b]	CO_2e kg/kg bonefree product
Beef	−0.538	−0.161	0.083	18.2	36[c]
Pork	−0.082	−0.370	−0.120	22.0	7.6[c]
Chicken	0.130	−0.518	−0.363	14.6	3.7[c]

[a]Säll and Gren (2015).
[b]Säll and Gren (2015) with the following factors for conversion to bonefree; beef 0.70, pork 0.59, chicken 0.77.
[c]Moberg et al. (forthcoming) using the same conversion factors as in b.

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II

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POLICIES AND SUPPORT IN RELATION TO LCA

Determining the climate impact of food for use in a climate tax—design of a consistent and transparent model

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Abstract

Purpose The aim of this study was to determine transparent food carbon footprint values for use in a climate tax, using a consistent methodology across the taxed food products and taking into account the need for such a tax to be administratively simple and accepted by affected stakeholders.

Methods A method based on Life Cycle Assessment following the ISO 14067 standard was developed for establishing simplified, yet consistent and transparent, datasets on the carbon footprint of food, for use as a base in a climate tax on food. Several sensitivity analyses were carried out to test the effects of inevitable methodological choices on the carbon footprint of different foods. The choices were then discussed in relation to taxation of food. The methodological choices included in the sensitivity analyses were different approaches to system boundaries, how to account for soil carbon changes and how to weigh greenhouse gases (GHGs).

Results and discussion The results on the carbon footprint of food calculated with the suggested method are in line with earlier findings in the field, with animal products, especially beef, showing a substantially higher value than most plant-based foods. Regarding choice of system boundaries for using the values in a tax, it is of particular importance to target emissions from biological processes, as these are currently untaxed. This would also be administratively simpler but less acceptable as large emission sources especially for imported products and greenhouse grown vegetables would not be included. Modelling emissions from soil carbon changes using a site-dependent method can be an advantage to obtain results in line with empirical data. Using Global Warming Potential over 100 years to weigh GHGs would be most in line with current climate reporting, which is an advantage for the consistency and acceptability of a tax.

Conclusions Ultimately, how taxes are set is a political decision, but food carbon footprint values determined with a consistent and simplified methodology are required in the process. This study presents carbon footprint values established using such method and provides valuable insights into how methodological choices affect the results of climate impact values and the implications for taxation.

Keywords Carbon footprint · Climate tax · Food consumption · Life Cycle Assessment

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1 Introduction

The current food system is a major contributor to climate change, causing 19–29% of total global greenhouse gas (GHG) emissions (Vermeulen et al. 2012). Animal-based products generally have a substantially larger climate impact than plant-based foods (Nijdam et al. 2012; Clune et al. 2017), so shifting away from animal-based products has been identified as an effective approach to achieve significant reductions in GHG emissions from the food sector (Springmann et al. 2018; Bryngelsson et al. 2016). However, changing people's consumption patterns is challenging (Hartmann and Siegrist 2017) and voluntary actions alone are probably not sufficient to reach climate targets (Garnett et al. 2015). Public financial policy, specifically a climate tax on food consumption, has therefore been proposed by non-government organisations (NGOs) and public authorities in countries such as Sweden (Lööv et al. 2013; SSNC 2015). Effects of such a tax have also been modelled in the scientific literature. For example, Säll and Gren (2015) found that emissions from the Swedish livestock sector could be reduced by 12% with a Swedish tax on animal products, while modelling of global pricing of food by Springmann et al. (2016) showed potential to reduce global food-related GHGs by 9%. A tax on consumption rather than production has been promoted, as such a tax has the advantage that domestic and imported products are equally affected, hence reducing the risk of emission leakage from moving production to other countries (Säll and Gren 2015; Van Doorslaer et al. 2015).

A climate tax on food consumption would be an example of an excise duty which Sweden, amongst other countries, have implemented already in several areas, e.g. on emissions of carbon dioxide (CO_2) from energy use and on tobacco and alcohol. Similar to how current excise duties are managed, an excise tax on food would probably be implemented in a tax system separate to other consumption taxes such as the value-added tax (VAT) (Swedish Tax Agency n.d.). In theory, an ideal cost-efficient tax on GHG emissions would be set at the marginal damage costs associated with emissions per unit (Pigou 1920). With respect to climate taxing food consumption, the tax on certain food products would then be determined by (1) the amount of emissions caused by the production and use of the food and (2) the marginal damage cost of the emissions. In this study, we investigated aspects involved in determining point (1), i.e. determining the carbon footprint of the food. The carbon footprint of a product is synonymous with a Life Cycle Assessment (LCA) that takes no other environmental impact category but climate change into consideration (ISO 2013).

Establishing the carbon footprint of foods involves a number of methodological choices, which can have a large impact on the output (Notarnicola et al. 2017). Therefore, before the implementation of a climate tax, supporting data on the climate impact of different foods must be established using a consistent methodology across all taxed food products. The datasets of the climate impact should reflect as correctly as possible the actual emissions from the life cycle of the food products on the Swedish market, in order for the tax to be cost-efficient and accepted by affected stakeholders. At the same time, the method used to establish the datasets has to be transparent in order to be administratively simple and easy to update and apply.

There is a large body of literature on the carbon footprint of food, ranging from assessments of specific food products to comprehensive reviews (Clune et al. 2017; Nijdam et al. 2012). Further, a number of databases exist that compile data on the climate impact of food on the global market, for example the World Food LCA database (Nemecek et al. 2015) and the Agri-footprint database (Durlinger et al. 2017), and on national level, for example the French AGRIBALYSE (Koch and Salou 2016) and the Swedish RISE database (Florén et al. 2015). However, none of these offer datasets suitable for being used in taxation, since no database or data in review studies is simultaneously (1) consistent in the methodology used across all food products, (2) representative for foods sold on the Swedish market and (3) sufficiently transparent.

The aim of the present study was therefore to determine transparent food carbon footprint values for use in a climate tax, using a consistent methodology across the taxed food products and taking into account the need for a tax to be administratively simple and accepted by affected stakeholders. To this end, a model for establishing consistent and transparent datasets on the climate impact of foods, to use as a base for taxation, was developed. To test how different unavoidable methodological choices affected the estimated climate impact of foods, various sensitivity analyses were conducted. Further, the implications of the choices for the values to be used in taxation were discussed, i.e. how the choices affected the criterion of implementing a cost-efficient, accepted and administratively feasible tax.

2 Background

2.1 Greenhouse gas emissions from the food system and current taxation on these

In comparison with the transport and energy sectors where emissions of GHGs are dominated by fossil fuel related CO_2, the emissions generated during the life cycle of foods include substantial amounts of methane (CH_4) and nitrous oxide (N_2O). Emissions of CH_4 mainly arise from feed digestion in ruminants, while the majority of the N_2O emitted originates from fertilised soils. Another important source of food-related GHGs is emissions of CO_2 due to soil carbon changes resulting from changes in land management and land use change (LUC), i.e. transformation of land from one use to another (Goglio et al. 2015). Additional emissions arise in the production of inputs to agriculture and in processing, refrigeration, packaging, storage and transportation of food commodities. In addition, use of refrigerants, e.g. for storing wild-caught fish on vessels, causes emissions of the hydrochlorofluorocarbon R22 (HCFC-22) (Ziegler et al. 2013).

Carbon taxes or emission trading systems (ETS) covering 13% of global GHG emissions have already been implemented in 40 countries, including Sweden (Författningssamling 1994; World Bank et al. 2016). These carbon-pricing mechanisms cover some of the CO_2 emissions associated with food production, namely those from the use of electricity and fuels. However, emissions of CH_4 and N_2O are currently untaxed except for N_2O emissions from mineral fertiliser production in the European Union (EU), which are included in the EU ETS. The compound HCFC-22 is not subject to any tax, but its use is now banned under an EU directive (European Parliament and Cote 2014) and it is thus currently being phased out. To the best of our knowledge, specific climate taxes on food have not yet been levied in any country.

2.2 Conditions for optimal tax levels

To obtain an ideal cost-efficient tax on GHG emissions, all emissions, irrespective of source, should be priced at the same level, as mitigation options will then be implemented in sectors where emissions can be reduced at the lowest cost (e.g. Baumol and Oates 1988). Therefore, taxes should ideally as closely as possible reflect the varying emissions generated when producing different products using different technologies. In the case of food, this would include differentiating between, e.g. vegetables grown in energy-demanding heated greenhouses and those grown in open fields and between different energy sources (e.g. fossil or renewable energy), in order to correctly reflect the true emissions from production. In addition, double taxation should be avoided by considering existing carbon taxes from the use of electricity and fuels in food production when establishing a tax on different food groups (Gren et al. 2019).

However, due to the high number of different food products available on the globalised food market of industrialised countries and the rapid development of new products, it is unlikely that a climate tax on food in practice could be based on detailed LCAs on each and every food product on the market. The cost of calculating and verifying the climate impact of all individual products would be unreasonably high (e.g. The Guardian 2012). For the same reasons, it is unlikely that differentiated charges for different technologies used during production can be implemented. Although it is possible to use national statistics to establish national averages for different foods from different countries, differentiating taxes based on country of origin of the product would not be a viable option, as that would likely violate the 'most-favoured nation' principle of the World Trade Organization (WTO). This principle states that members of the WTO must apply the same trading rules to all other WTO members, so differentiated tax rates for different countries would risk being discriminatory (Bähr 2015). Based on this, it would probably be necessary to base a tax on broader, more aggregated food groups.

It is important to also note that excise duties seldom are set so that they reflect the true externalities. Rather, they are a result of political negotiations including other considerations of the tax, such as being administratively simple (Government of Sweden 2009).

3 Methods

3.1 General model choices and input data used in the modelling

The proposed method uses a top-down approach to account for the GHG emissions associated with production of food from different countries available on the Swedish market. These are then aggregated based on import statistics to establish an average value for food sold on the Swedish market (Fig. 1). This approach enables the use of primary site-specific data for the most influential parameters in the calculations, such as yield levels and slaughter statistics, which are easily available in the form of datasets on national level for many countries (e.g. in national official statistics, National Inventory Reporting or national guidelines and reports from advisory services) (Fig. 2). This means that input data on the most influential parameters can easily be updated on a regular basis and that the results reflect the average carbon footprint of that food group on the Swedish market.

While primary site-specific data are often readily available for Sweden and many European countries, there are also cases when such data is not available for certain countries or food groups, e.g. fruits imported from South America. Data is then collected by the following order of prioritisation based on availability: data from the World Food LCA Database (Nemecek et al. 2015), peer-reviewed LCA studies and LCA reports. The World Food LCA Database is considered suitable, as it is available through the Ecoinvent database version 3.5 (Ecoinvent Centre 2018) and covers a variety of food products on the global market and offers access to the input data. Note however that using these data sources is a fallback solution that is used for food groups where primary data is not available. With future improvements in data availability, this can easily be changed.

For emission sources that are of less importance to the final results (e.g. packaging and electricity use in processing), standard values based on both primary and secondary literature data sources (carefully chosen to be representative for foods on the Swedish market) are used, which can simplify maintenance of the datasets. To further simplify maintenance, processes making a minor contribution to the final results (seed and seedling production, pesticide and mineral phosphorus and potassium fertiliser production, energy use for storage at wholesaler and retailer) are excluded. All input data to the model and further justifications for data choices are presented in the Electronic Supplementary Material.

3.2 Standardisation applied in the assessment

To align the method as far as possible with existing efforts to make LCA calculations consistent in the LCA community, thereby maximising acceptability and trust, the method follows the ISO 14067 carbon footprint standard (ISO 2013) for everything

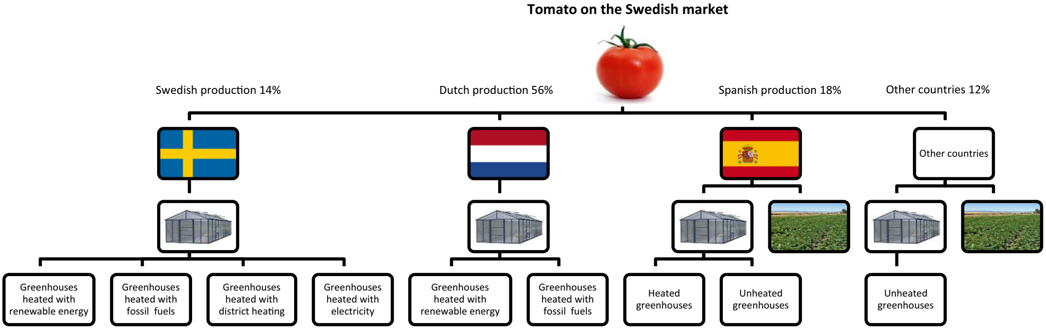

Fig. 1 Illustration of how the carbon footprint of a tomato on the Swedish market is assessed, i.e. aggregated from different countries using different production techniques

except the exclusion of biogenic carbon uptake, as this carbon is released to the atmosphere when the food is consumed.

There have been several initiatives to standardise calculation of the climate impact of products and foods in particular (e.g. BSI 2008, 2012; Food SCP RT 2013; European Commission 2016). However, none of the available standards are sufficiently specific and detailed to be used in the present application without modification. For example, most existing standards are based on the use of product category rules (PCR), which are sets of specific rules applied to different products (e.g. PCRs for beer, olive oil and meat have been developed within the EU's product environmental footprint (PEF) initiative (European Commission 2016)). For taxation, a method is needed that is consistent *across* food groups, not just within food groups.

3.3 Methodological choices in the modelling

3.3.1 Attributional LCA modelling

The method uses an attributional LCA (ALCA) approach, where GHG emissions directly associated with the production steps are accounted for.

Another approach in LCA is consequential LCA (CLCA), which accounts for the change in environmental impact as a consequence of a changing market using marginal data (Finnveden et al. 2009). With the CLCA approach, emissions caused by producing a specific food item on the margin are calculated, which could be considered more in line with economic theory on taxing marginal emissions. However, such an approach would require knowledge of the marginal technologies in food production, i.e. where, and with what technologies, the production of a certain food group would increase or decrease as a consequence of a marginal market change. This is possible to estimate using, e.g. economic equilibrium models (e.g. Kløverpris et al. 2010), but the outcomes of these models depend on many assumptions and are therefore highly uncertain, difficult for non-experts to interpret and costly to update.

Therefore, as Brandão et al. (2014) suggest, ALCA methodology can be considered more suitable for use in policy for the *implementation* of a decision to move towards a policy goal such as changed consumption patterns of food.

3.3.2 Reference unit

With the proposed method, the climate impact is assessed based on the reference unit of 1 kg of product (litre for drinks and oils).

Here we avoid the use of the term 'functional unit' not to give the impression that different food groups are functionally equal. Comparing foods per unit mass in LCA has been criticised for failing to consider the function of foods and alternative ways to account for the function have therefore been suggested (Notarnicola et al. 2017). Alternatives are commonly based on the nutritional quality of the foods, e.g. by use of energy content (kilocalories), or single nutrients, e.g. kg of protein, in the reference unit. Such units may be suitable when comparing the climate impact within one food category providing the same function (e.g. using per kg of protein for protein-rich foods such as meat, dairy, egg, legumes and cereals), but is less suitable when comparing products from different food groups. For example, using energy content as the reference unit would not capture the benefits of fruit and vegetables, which are low in energy but dense in fibre and important micronutrients.

Instead, to reflect the overall nutritional quality of foods, a 'nutrient index' can be used (Hallström et al. 2018). The basis of most nutrient indices is to consider the content of a range of nutrients in food products in relation to the recommended daily intake of these nutrients. Several models have been developed which differ, e.g. with respect to the nutrients included, daily recommended values and the algorithm used to compute the index (Drewnowski 2009). Although it would be possible to relate the climate impact of foods to their nutritional quality, the design of nutrient indices and the combined values would rely on many choices and limit the transparency of datasets, which potentially could reduce understanding and acceptance of a tax.

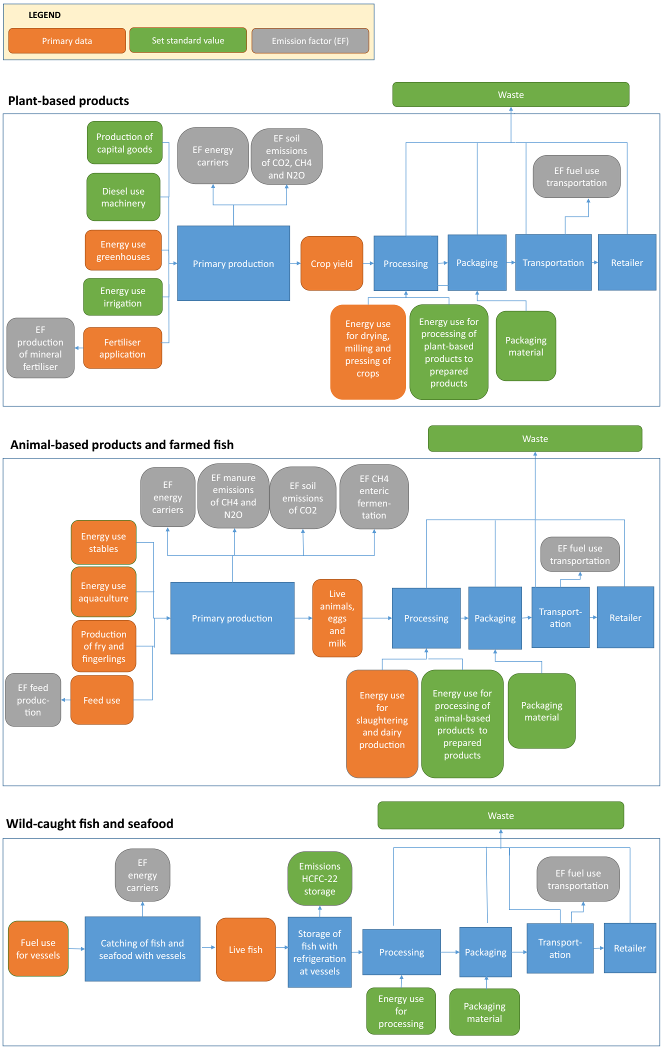

Fig. 2 Conceptual model of the method suggested for calculating the carbon footprint of foods on the Swedish market for use in a climate tax of food

Further, using a joint climate-nutrition dataset of foods for taxation would limit the cost efficiency of a tax, as there is no clear relationship to the damage cost of these aspects combined. Rather, using 1 kg as the reference unit is likely the only valid reference unit on which to base a tax as the emissions to be taxed should be directly related to the amount of food purchased.

3.3.3 Allocation method

Economic allocation is used in the proposed method to allocate emissions from joint production systems across different products (e.g. milk and meat from dairy production).

Other strategies are possible, such as physical allocation based e.g. on mass or energy content of each by-product. Physical allocation is used in the Renewable Energy Directive (RED) of the EU (European Union 2009), based on the energy content in fuels. This is a viable option for the specific case of fuels, but when considering taxes for different categories of food providing different functions, it is difficult to decide upon a suitable physical property (compare discussion of the reference unit in the 'Reference unit' section). Instead, economic allocation can be considered more suitable as it can be used across products, which is important for consistency in the methodology. Due to fluctuating market prices, economic allocation factors should preferably be calculated as an average of a longer time period in order to be robust.

3.3.4 System boundary

Our proposed method uses a system boundary from cradle to Swedish retail, i.e. accounting for emissions arising from the production of input materials, primary production, processing, packaging, transportation and food losses in the different stages until the food is available on the Swedish market (losses are included for the retail stage too) (Fig. 3).

The full life cycle of a food product also includes processes after retail, but many of these post-retail emissions are already covered by the Swedish CO_2 tax (Fig. 3). It can therefore be argued that these emissions should be excluded from a climate tax on food, as double taxation of products should be avoided for a Pigovian tax to be cost-efficient (Gren et al. 2019). The same applies to energy use in processing, packaging, storage and transport, as these are covered by the national CO_2 tax, although there are partial reductions in the tax for some sectors. On the other hand, international transport is not always taxed, certainly not sea or air transport, and energy use in food production or production of packaging material abroad may not be. Hence, we choose to include these life cycle steps in the tax scheme in order to reflect the full carbon footprint up to retailer.

How to administer a tax, especially the point in the food system at which the tax should be paid, may also influence where and how system boundaries are drawn. Current excise taxes, e.g. on tobacco and alcohol, are paid by the party that produces or imports the product, so supermarkets, restaurants, etc. buy the already taxed product (Swedish Tax Agency n.d.). Hence, tax payments are made by fewer companies and are therefore easier to control and administer (Statistics Sweden 2018). Applying the same approach to the food sector would require taxes to be paid by major producers of food commodities, e.g. dairy companies, mills and slaughterhouses (Fig. 3). With such a system, a cradle-to-farm gate approach would be more suitable for a climate tax on food. The post-farm gate emissions, which are mainly energy-related, would then need to be covered by a CO_2 tax on energy, which is already in place in Sweden and many other European countries.

Another, simpler, approach could be to limit the tax to emissions from the agricultural sector only as defined by the IPCC (Smith et al. 2014), i.e. including only emissions from biological processes giving rise to emissions of CO_2, CH_4 and N_2O (emissions from soils, enteric fermentation and manure management). These emissions are currently untaxed, while energy and transport-related emissions are (or should be) covered by CO_2 taxes. To test how the choice of system boundary affected the modelling outcome, a sensitivity analysis was carried out using different approaches (see 'Discussion' section).

3.3.5 Accounting for carbon changes in soils

Our proposed method includes soil carbon changes by using a simplified strategy based on the Introductory Carbon Balance Model (ICBM) (Andrén and Kätterer 1997).

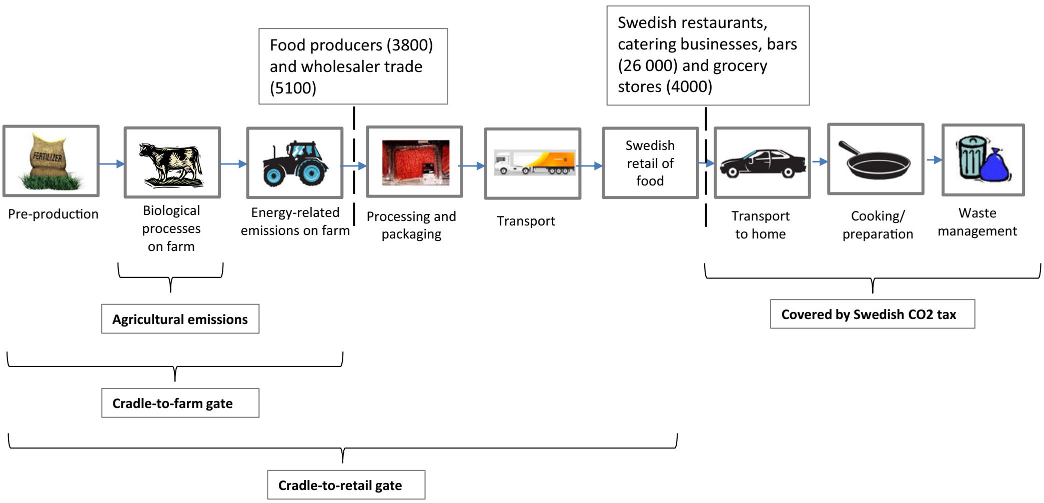

Fig. 3 System boundaries according to the point of administration of a tax, with approximate number of actors in each stage in brackets. Data retrieved from Statistics Sweden (2018) and Svensk Dagligvaruhandel (2018). Pictures used with permission from Viktor Wrange and Fredrik Saarkoppel

Changes in soil carbon can have a large impact on the climate impact of food and other biomass-based products. Emissions due to soil carbon changes are currently untaxed and sequestration of carbon in soils is not financially rewarded in Sweden. Soil carbon changes are included in other policies, e.g. the RED (European Union 2009), and schemes to pay farmers for the carbon sequestered in soils have been implemented in countries such as Australia (Australian Government n.d.).

According to the ISO 14067 standard, changes in soil carbon should be included in carbon footprint assessments based on an internationally recognised method such as the IPCC Guidelines for National Greenhouse Gas Inventories (NIR) or on a national approach based on a verified study (ISO 2013). The ICBM is used in the Swedish NIR (SEPA 2017) and accounts for the emissions/sequestration in mineral soils based on the driving variables of climate and carbon input to soils from, e.g. manure and crop residues. We used this method to assess the 'soil carbon change potential', i.e. the amount the soil would sequester or loose until it reaches steady state, from cultivation of cereals, legumes, oilseed rape, root vegetables and grass-clover ley. Cultivation of these crops leads to very different carbon inputs to soil (carbon input being a main determinant of soil carbon changes). This potential change was compared with the current average carbon content in Swedish soils as a reference (100 t C per ha; Poeplau et al. 2015). Further, the change was annualised over 100 years and divided by the annual yield, from which an estimate was obtained of the potential annual CO_2 emissions/sequestration caused by soil carbon changes per kg of crop (Electronic Supplementary Material). As the ICBM is mostly used in Sweden, a more general approach to account for soil carbon changes (IPCC 2006) was tested in a sensitivity analysis ('Accounting for carbon changes in soils' section).

3.3.6 Accounting for emissions due to land use change

LUC emissions for soybean and oil palm in animal feed are included in the proposed method using emissions factors from Henders et al. (2015).

Emissions from LUC arise when land is transformed from one use to another, e.g. from forest to cropland (Edenhofer et al. 2011), and can contribute considerably to the carbon footprint of food (e.g. Flysjö et al. 2012). Deforestation in South America and Asia, driven by exports of soybean and palm oil to Europe for use as animal feed, is a highly debated issue in Sweden and the European food sector (e.g. WWF 2014). As these emissions are currently untaxed, they should be included in a climate tax on food. The ISO 14067 standard states that emissions from LUC should be accounted for in a product's carbon footprint if they arise from a change in land management within a studied product system, i.e. from direct land use change (dLUC) (ISO 2013). Such emissions are normally allocated to the crops grown on the converted land (Goglio et al. 2015). As opposed to dLUC, indirect land use change (iLUC) describes how an increase in production of a certain crop displaces other crops in an area. To keep up with demand for the displaced crop, production is moved to locations for which new and previously untilled land is cleared, causing GHG emissions (Röös and Nylinder 2013).

Various methods for accounting for emissions from LUC are available but give highly variable results, as the assessment, especially for iLUC, relies on several assumptions about the drivers of LUC (Finkbeiner 2014; Persson et al. 2014). Based on this, ISO 14067 states that iLUC emissions should be considered in climate impact assessments once international consensus has been reached (ISO 2013). However, none of the methods currently available for determining either dLUC or iLUC are generally accepted, and reaching consensus on methods will probably take considerable time, or may not even be possible, as choice of method reflects different perspectives on the problem (Flysjö et al. 2012).

Although the uncertainties in LUC calculations could favour exclusion of these emissions from a climate tax on food, we argue that for acceptance of a climate tax on food and because LUC emissions are currently untaxed, it is important to include the emissions in a tax. The LUC factors from Henders et al. (2015) are based on a method suggested by Persson et al. (2014) but updated with more recent data (Electronic Supplementary Material). The method takes both dLUC and iLUC into account by calculating the average emissions caused by LUC for a certain agricultural commodity and region (e.g. soybean from Brazil or palm oil from Indonesia), rather than allocating LUC emissions only to crops grown on recently deforested land. This method, apart from representing the latest developments in accounting for emissions from LUC, has an important advantage when using it for a tax; as LUC emissions are calculated on the 'average' commodity from a certain region, not just the crops grown on recently deforested land, it can be applied equally to all commodities from a certain region. That means that, e.g. all soybean from Brazil has the same LUC factor regardless of whether it is grown on newly deforested land or not (as long as deforestation driven by soybean cultivation continues in Brazil).

3.3.7 Weighting emissions and tax levels of CO_2, CH_4 and N_2O

To weigh the impact of different GHGs into one common unit, we use the Global Warming Potential over 100 years (GWP_{100}) according to the latest IPCC report (Myhre et al. 2013).

Using GWP_{100} is also the recommendation of the ISO 14067 standard (ISO 2013). The factors for GWP_{100} are available both with and without the effects of climate-carbon feedback mechanisms (Myhre et al. 2013). These take into consideration the effects of climate impact on changes in the climate cycle that can further amplify (or dampen) climate change. In our proposed method, the feedbacks are included based on the recommendations by UNEP/SETAC Life Cycle Initiative (2016). For CH_4, N_2O and other non-CO_2 GHGs, there is generally greater uncertainty in the metrics than for CO_2 when including the feedback mechanisms (Levasseur et al. 2016). To test the sensitivity of including the feedback mechanisms, as well as of using metrics other than the GWP_{100} and taxing each gas separately, a sensitivity analysis was carried out (section 5.3).

3.3.8 Summary of methodology used in the proposed method

A summary of the methodology used in the proposed method is presented in Table 1. Factors for all methodological choices and those tested in the sensitivity analysis are presented in the Electronic Supplementary Material.

4 Results

The carbon footprint of a set of food groups on the Swedish market, calculated using the proposed method, is presented in Fig. 4. Results for other food groups are provided in the Electronic Supplementary Material, together with transparent results for all products divided over separate processes in the life cycle stages. The results were validated against existing datasets and studies; Fig. 4 provides a comparison with the values (median and variation intervals) in Clune et al. (2017) which is a compilation of published LCA studies on different foods, i.e. they come from different countries and are performed using different methodologies. The results are in line with the results reported by Clune et al. (2017), with plant-based foods and especially fruit and vegetables having a substantially lower carbon footprint than animal products, particularly ruminant meat. Using the method presented here gives higher carbon footprint for beef and dairy products than the median value in Clune et al. (2017) due to the use of the latest (and higher) characterisation factor for CH_4 (34 instead of 25). The higher value here for grains is explained by accounting for waste along the chain and at retailer, which also partly explains why the carbon footprint for fruit and vegetables on the Swedish market is higher than the median values in Clune et al. (2017). For fruit and vegetables, longer transport distances also add to the discrepancy between the carbon footprints calculated here and the data in Clune et al. (2017).

5 Discussion

The simplified top-down method proposed here to calculate the carbon footprint of foods on the Swedish market showed to produce results in line with previous studies. At the same time, it fulfils the aim of being useful and robust for use for a climate tax on food in the following ways:

- The method is *consistent* as the same methodological choices are used across all food groups included.
- It produces carbon footprints of foods which is *representative* of foods on the Swedish market by accounting for food imports and production methods in different countries.
- The data and methodological choices are *transparent*, the method builds on publically available data and the full model including all data is made available.

However, in the development of this method, several unavoidable methodological choices had to be made. Therefore, in the 'Choice of system boundaries', 'Accounting for carbon changes in soils' and 'Weighting emissions and tax levels of CO_2, CH_4 and N_2O' sections, sensitivity analyses of the most important choices are presented to provide further justification of these choices. In the 'Limitations' section, general limitations of the method are discussed.

5.1 Choice of system boundaries

Figure 5 shows how the choice of system boundary affects the climate impact of an illustrative set of foods, using the system boundaries in Fig. 3 (for all food groups, see Electronic Supplementary Material). Limiting a climate tax on food to agricultural emissions of CO_2, CH_4 and N_2O would exclude on average 46% of emissions for pork and 50% for chicken but only 18% for beef and 32% for milk (see Electronic Supplementary Material). Emissions from agriculture also dominate the climate impact of rice, due to high emissions of CH_4 from flooded rice paddy fields. For plant-based foods

Table 1 Summary of the methodology used in the proposed method

Type of LCA	Reference unit	Allocation method	System boundary	Changes in soil carbon	Emissions due to LUC	Weighting of GHGs
ALCA	Per kg[a] or per litre	Economic	Cradle to retail gate (including waste through all stages and at retailer)	Included based on the ICBM over 100 years (Andrén and Kätterer 1997)	Included based on method suggested by Persson et al. (2014)	GWP_{100} with climate-carbon feedbacks (Myhre et al. 2013)

[a] Per kg bone free weight for meat, per kg edible weight for fish and seafood

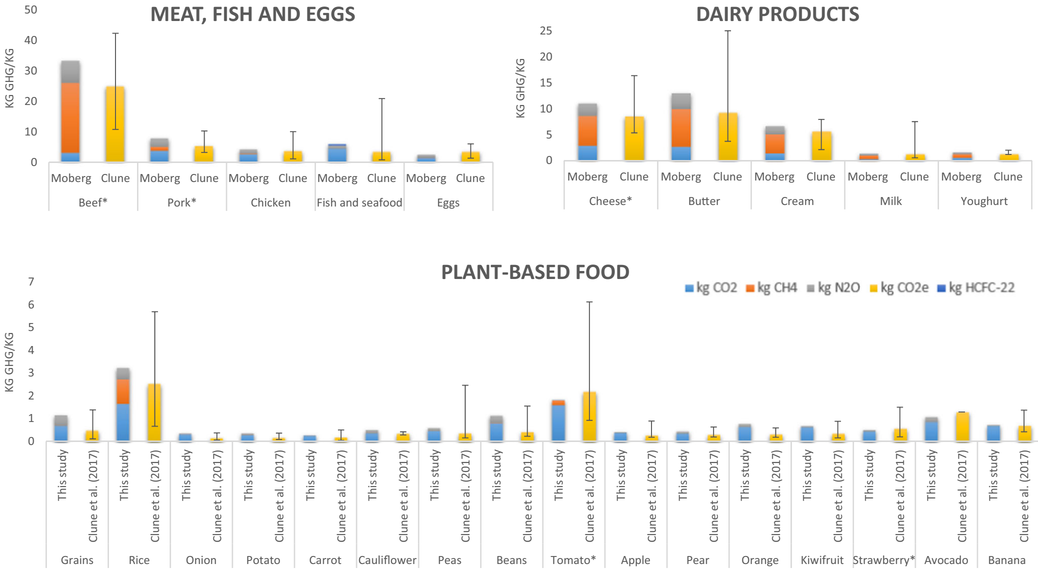

Fig. 4 Climate impact per kg of food (per kg bone free weight for meat and edible weight for fish) on the Swedish market from this study, validated to median values and variation intervals in Clune et al. (2017). Beef and pork*: EU Average from Clune et al. (2017). Cheese*: average of hard and dessert cheese. Tomato*: only heated greenhouse. Strawberry*: only open-field

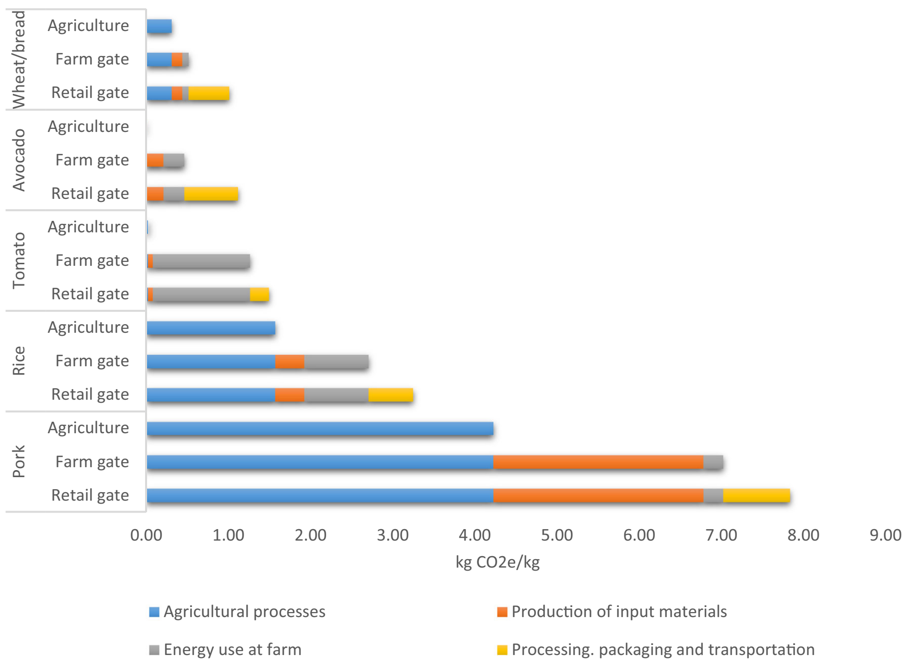

Fig. 5 Carbon footprint of a set of food products using different system boundaries

grown in greenhouses on the other hand, e.g. tomatoes, emissions from energy use are a major contributor in the life cycle. For tomatoes, taxing only agricultural emissions or full cradle-to-retail emissions would result in a very different tax.

As mentioned, taxing only agricultural emissions would probably be the easiest option with respect to administration. In addition, it would target emissions that are currently untaxed and avoid double taxation of products. However, it would risk excluding many of the emissions from e.g. plant-based foods, such as packaging, processing and transportation, which might lead to less understanding and acceptance of a tax as these processes are often perceived by consumers to have great influence on the overall climate impact (Shi et al. 2018). Thus, choosing to tax either up to farm gate or retail gate might gain more understanding and acceptance of a tax. Using set values for processes after farm gate, as suggested here ('General model choices and input data used in the modelling' section), would ease the administration and implementation of a tax from cradle to retailer.

In conclusion, the choice of system boundary for a tax involves a trade-off between ease of administration and accuracy of emission levels, where the latter is important for cost efficiency and acceptance.

5.2 Accounting for carbon changes in soils

The ICBM model used here to assess the 'soil carbon change potential' (section 3.3.5) requires input of regional data, which is lacking for many food production systems outside Sweden. The Tier 1 approach of the IPCC NIR guidelines for assessing soil carbon changes, on the other hand, is based on fixed factors for different levels of land management, tillage intensity and inputs of organic material (IPCC 2006). As the IPCC method is very simple and could be used consistently for foods from different world regions, we tested if this would be a suitable method for accounting for soil carbon changes. As in the ICBM approach, the 'soil carbon change potential', annualised over 100 years, was assessed from cultivation of 1 kg of cereals, legumes, oilseed rape, root vegetables and grass-clover ley. Figure 6 shows the results of the two different approaches for an illustrative set of food groups.

For ruminant products, the climate impact was expected to decrease due to the large proportion of grass in ruminant diets. The results from the ICBM modelling followed this assumption, with a decrease in the overall impact of beef by 5% and milk by 3% (Fig. 6). When using the IPCC Tier 1 approach however, the climate impact of both products instead increased (Fig. 6), as the emission rates for annual crops used in ruminant diets exceeded the sequestration rate of ley (Electronic Supplementary Material). Hence, the IPCC approach failed to capture the sequestration potential of Swedish grass-clover leys that is well described in the literature (Poeplau et al. 2015), while the ICBM approach gave results in line with empirical data for the Nordic countries (Kätterer et al. 2013). Hence, using the ICBM approach is a better choice in this case, although it might not correctly reflect the soil carbon changes associated with imported products. Due to the uncertainties in accounting for soil carbon changes, it could be argued that these emissions should be excluded in a climate tax on food. However, as accounting for soil carbon changes may substantially affect the carbon footprint of foods, we argue that including the emissions or sequestration in an uncertain way with average values is more accurate than excluding them entirely. As LCA studies neglecting to account for soil carbon have been criticised repeatedly by researchers (e.g. Stanley et al. 2018) and by proponents of grass-fed ruminant production (e.g. Rundgren and Bremen 2017), including soil carbon changes can be especially important for acceptance of a climate tax on food, as it decreases the difference between ruminant and mono-gastric meat.

A general limitation of both models is that it is far from certain that the emissions or sequestration of carbon attributed to different crops will actually take place (compared with CH_4 emissions from enteric fermentation, which are known to take place), as the methods are based on average regional conditions rather than site-specific conditions. For example, much ruminant production takes place on soils that are already high in carbon, as cropping systems on these farms have been dominated by ley for a long time, and these soils therefore have limited potential for sequestering additional carbon. Conversely, many annual crops are cultivated on soils less rich in carbon and these soils would not emit as much carbon during cropping as 'average' carbon-rich soils.

5.3 Weighting emissions and tax levels of CO_2, CH_4 and N_2O

As previously mentioned, excise taxes should in theory be designed according to the marginal damage and associated cost of the climate impact. The marginal damage cost can be set on separate GHGs (one price for CO_2 emissions, one for CH_4 emissions and so on) or on CO_2e based on e.g. GWP_{100} (Marten et al. 2015). Figure 7 shows the marginal damage cost using different approaches for weighting different GHGs: GWP_{100}, the Global Temperature Potential for 100 years (GTP_{100}) (Myhre et al. 2013), i.e. based on the temperature change caused by the different gases in 100 years, and GTP factors based on Persson et al. (2015). Persson et al. (2015) argue that using a 100-year time horizon for GTP is not in line with the Paris agreement target to limit warming to 2 °C, as this warming is expected to be reached within 100 years (at some point between 2050 and 2100). They therefore suggest using GTP corresponding to times when the target is expected to be met and to account for uncertainty in these timings. For GWP_{100} and GTP_{100}, the cost both with and without inclusion of climate-carbon feedbacks for CH_4 and N_2O is shown. Further, the marginal damage cost of separate GHGs based on the cost estimates by Marten et al. (2015) is applied.

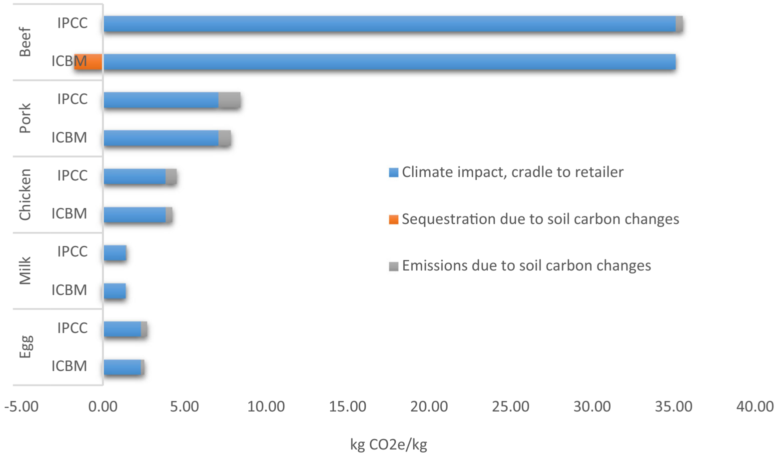

Fig. 6 Carbon footprint of a set of food products, with emissions and sequestration due to soil carbon changes calculated with the ICBM approach and with the IPCC Tier 1 approach

Differences in how the GHG emissions were weighted were most pronounced for beef, cheese and rice, whereas for other products such as potato, the different weighting approaches had minor effects. This is due to the large emissions of CH_4 that arise from the production of these products and to the difference between the characterisation factors used for CH_4. With the exception of rice, non-ruminant products generally cause more emissions of CO_2 and N_2O, for which weightings according to different characterisation factors were quite similar (Electronic Supplementary Material). Including climate-carbon feedback increased the climate impact for all products, but the differences were more pronounced for products causing large emissions of CH_4 (Fig. 7).

Implementing a tax based on the marginal damage cost of each separate GHG is argued to be most cost-efficient as an optimal climate tax should be differentiated between GHGs due to their different climate impact (Gren et al. 2019). Current climate reporting is however based on GWP_{100}, why using GWP_{100} would be an advantage for acceptance and ease of administration. And as the difference between using GWP_{100} in taxation and taxing GHGs separately is small in most cases, as shown in Fig. 7, we choose to use GWP_{100} in the suggested method.

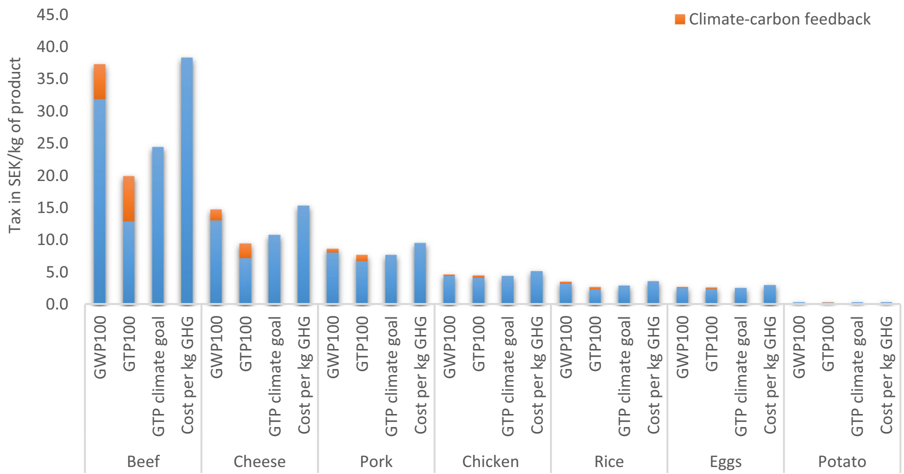

Fig. 7 Tax rates in SEK with different weighting of the climate impact and with/without inclusion of climate-carbon feedbacks. A social marginal cost is applied of 1120 SEK/t for CO_2, 35,797 SEK/t for CH_4 and 438,762 SEK/t for N_2O, according to Marten et al. (2015). Prices in USD in Marten et al. (2015) are converted using the 2017 exchange rate

Both GWP and GTP rely on an arbitrary choice of time horizon, often fixed to 100 years, which can affect climate mitigation priorities. Using GTP factors as suggested by Persson et al. (2015) would be a more accurate option in order for a tax to move towards an actual climate goal. However, using GWP_{100} for a climate tax would be more consistent with how climate policy is developed to date. Including climate-carbon feedbacks for all GHGs and not only CO_2 in GWP factors would introduce a trade-off between consistency in the methodology and accuracy, due to the uncertainty of the calculations. As recommended by UNEP/SETAC Life Cycle Initiative (2016), our proposed method includes climate-carbon feedbacks, for consistency in calculations.

5.4 Limitations

The method suggested here to calculate the climate impact values has several limitations. It could be critiqued both for being overly complicated, including too many (unimportant) processes and requiring too much input data, e.g. for packaging and transports, and for being overly simplified, e.g. not distinguishing between different processing or packaging alternatives. The method is the result of a balancing act between completeness in order to be acceptable and 'correct' and simplicity in calculations and maintenance. Ultimately, how taxes are set is the result of political negotiations and compromises, the final result not always reflecting what is 'optimal' or 'correct'. The transparent method presented here can however provide valuable input to such discussions.

As for data availability, this is also a challenge for certain production countries, food groups and production systems, especially as regards agricultural statistics and data on regions outside Europe. For food groups such as fish and seafood, we were unable to find any official data on key parameters such as fuel use and landed catch of fishing vessels. Hence, for some products, input data for the present study had to be taken from the inventories in earlier LCA studies. However, ongoing initiatives in compiling databases for fish and seafood may facilitate future assessment (e.g. Parker et al. 2018). Tracing food trade is also a challenge; trade statistics may hide the true country of origin, which hinders country-specific assessments for some food products and assumptions regarding the export country have to be made.

Regarding the maintenance of the dataset, this can also be a challenge considering the amount of data that is required. However, input data on the most influential parameters can easily be updated on a regular basis, as these come from reports that are compiled by authorities for other purposes. For emission sources where there is limited data availability or where set values are used, updates can be made less frequently. Detailed suggestions for maintenance of datasets are presented in the Electronic Supplementary Material.

6 Conclusions

This paper presents transparent food carbon footprint values for use in a climate tax, established using a consistent methodology across the taxed food products. A condition in determining the datasets was that the tax had to be administratively simple and accepted by affected stakeholders. The climate impact values were established by primarily using official national data, which facilitates data collection and updating of the values. The climate impact of foods available on the Swedish market calculated with the suggested model was in line with values reported in earlier studies in the field.

A sensitivity analysis on different approaches to setting system boundaries revealed that limiting a climate tax on food to agricultural emissions of CO_2, CH_4 and N_2O would target currently untaxed GHGs and avoid double taxation of CO_2 emissions from energy and fuel use. However, it would impose a trade-off between ease of administration and accuracy of emission levels of food at the retailer, where the latter is important for cost efficiency and acceptance. Despite uncertainties in accounting for soil carbon changes, including these emissions is important for consistency with calculation methodology and acceptance of a tax, as they substantially affect the carbon footprint of foods and especially the relative difference between animal-based products. Modelling emissions from soil carbon changes using a site-dependent method can be an advantage to obtain results in line with empirical data. For weighting different GHGs, taxing each gas individually would be more cost-efficient, but using GWP_{100} for a climate tax would be most in line with current climate reporting, which could improve acceptance and consistency.

Ultimately, how taxes are set is a political decision, but carbon footprint values of food determined using a consistent, simplified method are required in the process. This study presents values of the climate impact established with one such method and provides valuable insights into how methodological choices affect the carbon footprint values obtained and the implications for taxation. This is indispensable knowledge in the political process of establishing a climate tax on food.

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III

sustainability

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Article

Benchmarking the Swedish Diet Relative to Global and National Environmental Targets—Identification of Indicator Limitations and Data Gaps

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Abstract: To reduce environmental burdens from the food system, a shift towards environmentally sustainable diets is needed. In this study, the environmental impacts of the Swedish diet were benchmarked relative to global environmental boundaries suggested by the EAT-*Lancet* Commission. To identify local environmental concerns not captured by the global boundaries, relationships between the global EAT-*Lancet* variables and the national Swedish Environmental Objectives (SEOs) were analysed and additional indicators for missing aspects were identified. The results showed that the environmental impacts caused by the average Swedish diet exceeded the global boundaries for greenhouse gas emissions, cropland use and application of nutrients by two- to more than four-fold when the boundaries were scaled to per capita level. With regard to biodiversity, the impacts caused by the Swedish diet transgressed the boundary by six-fold. For freshwater use, the diet performed well within the boundary. Comparison of global and local indicators revealed that the EAT-*Lancet* variables covered many aspects included in the SEOs, but that these global indicators are not always of sufficiently fine resolution to capture local aspects of environmental sustainability, such as eutrophication impacts. To consider aspects and impact categories included in the SEO but not currently covered by the EAT-*Lancet* variables, such as chemical pollution and acidification, additional indicators and boundaries are needed. This requires better inventory data on e.g., pesticide use and improved traceability for imported foods.

Keywords: food consumption; environmentally sustainable diets; EAT-Lancet; Planetary Boundaries; Swedish Environmental Objectives; environmental indicators

1. Introduction

The food system is a major contributor to many environmental pressures, threatening the functioning of several Earth systems [1]. Food-related activities account for 19%–29% of global greenhouse gas emissions (GHGs) [2], occupy about 40% of the Earth's land surface [3] and are the main driver of deforestation of tropical forests, causing large GHG emissions and threatening biodiversity [4]. Moreover, the agriculture sector is responsible for 70% of global freshwater withdrawals and pollutes aquatic and terrestrial ecosystems through emissions of nitrogen and phosphorus from fertiliser use [1].

To reduce these environmental burdens, profound changes in the food system are needed, including a shift towards environmentally sustainable diets [1]. These are often regarded as diets causing lower environmental impacts (e.g., [5]), which several studies have identified as diets with a low to moderate amount of meat and animal-based products and a larger share of plant-based foods than in current diets in many high-income countries [6–9]. This conclusion was also reached by Martin and Brandão [10], who investigated implications of dietary changes by the Swedish population. The average Swedish diet involves high intake of meat and dairy products in comparison with both the European and global average [11,12], and Martin and Brandão [10] demonstrated that switching to a vegetarian or vegan diet could decrease several environmental burdens. However, although such results give valuable insights into the relative performance of diets, they do not show whether the diets are sustainable 'enough' to reach environmental targets. For this, it is necessary to evaluate the environmental performance of the diets relative to absolute thresholds, beyond which they can be considered unsustainable. Such thresholds are highly challenging to establish, but some attempts have been made. One example is the 'One Planet Plate' concept, where the World Wildlife Fund (WWF) in Sweden sets absolute limits for GHG emissions from yearly consumption of food [13]. Another example is given by Röös et al. [14], who investigated the environmental sustainability of the average Swedish diet by defining per capita thresholds for GHG emissions and occupation of agricultural land for Swedish food consumption, and then benchmarking the impacts of the diet against these boundaries. Furthermore, the EAT-*Lancet* Commission [1] recently proposed absolute boundaries for six Earth system processes, within which the global food system should operate to be environmentally sustainable.

As previous studies evaluating sustainable diets have primarily focused on GHG emissions and land use as indicators of sustainability [15,16], the boundaries proposed by the EAT-*Lancet* Commission make it possible to evaluate the absolute environmental sustainability of diets with a more comprehensive set of indicators than before. However, although the EAT-*Lancet* Commission proposes several environmental indicators, these do not capture all environmental sustainability issues. One such example is the use of pesticides, which can have toxic impacts on humans and ecosystems. Furthermore, the EAT-*Lancet* boundaries are defined on a global level, so applying them at national scale might risk overlooking important aspects. The indicator for GHG emissions is the only exception, since emissions of GHGs cause global problems, regardless of the source of the emissions. For other aspects such as freshwater use, the impacts have a strong local connection, i.e., where and how water is used is highly important [16]. Hence, national indicators may add perspectives on the conditions where the majority of the foods in the diet are produced. For Sweden, such national indicators are found in the framework of the Swedish Environmental Objectives (SEOs). These are intended to steer Sweden's environmental policy towards solving environmental issues for the next generation, without causing environmental problems outside Sweden's borders [17] (Summary in English available at: http://www.swedishepa.se/Documents/publikationer6400/978-91-620-8620-6.pdf).

The aim of the present study was to assess the environmental sustainability of the Swedish diet and evaluate how well global indicators capture local environmental sustainability concerns. This was achieved by assessing and benchmarking the environmental impacts of the Swedish diet against global environmental sustainability boundaries and identifying potential missing aspects and indicators for capturing the environmental sustainability of the diet in a local context.

2. Materials and Methods

We used the variables proposed by the EAT-*Lancet* Commission [1] as the starting point for evaluation of the environmental sustainability of the average Swedish diet. Absolute boundaries for the environmental sustainability of key Earth systems affected by global food production are proposed by the EAT-*Lancet* Commission, making it one of the most comprehensive frameworks for assessing food system environmental sustainability against absolute boundaries.

To calculate the impacts caused by the average Swedish diet, we retrieved data on average per capita food supply (Section 2.1), assessed environmental impacts per kg or litre of food and then multiplied the amounts of food by the environmental impacts. Inventory data used for assessment of the foods are described in Section 2.2 and are available with references, in the Supplementary Material. We then benchmarked the environmental impacts from the per capita average diet against the downscaled EAT-*Lancet* boundaries (Section 2.3) for each environmental variable. To identify local environmental concerns not captured by the EAT-*Lancet* variables, we analysed the relationships between the SEOs, which are designed for describing and monitoring the local environmental status in Sweden, and the global EAT-*Lancet* variables. Based on this, we looked for additional indicators relevant for capturing the environmental sustainability of diets in the local context (Section 2.4).

2.1. Data on Food Supply for the Average Swedish Diet

Data on food supply for the average Swedish diet were obtained from the Swedish Board of Agriculture [18,19]. We used data on the average direct consumption of food, i.e., the amount of food available for consumption. This food could be either eaten or wasted, but is the amount of food that needs to be produced to sustain the Swedish diet. A full list of the food supply is available in the Supplementary Material (Table S4). To reduce impacts from variations between years, an average of the food consumed in the years 2011 to 2015 was used. Lists of raw materials in processed and prepared products were obtained from the Swedish National Food Agency (as unpublished data on the amount of raw agricultural commodities (RAC)) [20], from Life Cycle Assessment (LCA) studies [21] and from reports [22,23].

2.2. Global Boundaries, Indicators and Inventory Data

The EAT-*Lancet* framework [1] (Table 1) builds on the 'Planetary Boundaries' concept [24,25], which defines absolute environmental sustainability limits for nine Earth system processes pressured by human activities. The majority of the boundaries in the EAT-*Lancet* framework are set in relation to the Planetary Boundaries framework, i.e., are based on absolute biophysical limits for Earth systems affected specifically by food production within which humanity should operate.

Table 1. Earth system processes, control variables and global food system boundaries defined by the EAT-*Lancet* Commission [1]. GHG = greenhouse gases, CO_2e = carbon dioxide equivalents, E/MSY = extinctions per million species-years. Range of uncertainty for the global boundaries is given in parentheses.

Earth System Process	Climate Change	Land-System Change	Nitrogen (N) Cycling	Phosphorus (P) Cycling	Freshwater Use	Biodiversity Loss
Control variable	GHG emissions	Cropland use	N application	P application	Consumptive water use	Extinction rate
Global boundary	5 Gton CO_2e per year (4.7–5.4) *	13 million km^2 (11–15)	90 Tg N per year (65–130)	8 Tg P per year (6–16)	2500 km^3 per year (1000–4000)	10 E/MSY (1–80)

* Represented by emissions of methane and nitrous oxide from agriculture and minor emissions of CO_2 from biomass burning. CO_2 emissions from fossil sources and land use change should be zero (see Section 2.2.1).

2.2.1. Climate Change

We evaluated the climate impact caused by the average Swedish diet by using the control variable of GHG emissions, as suggested by the EAT-*Lancet* Commission [1]. While the majority of the boundaries in the EAT-*Lancet* framework are set in relation to the absolute biophysical Planetary Boundaries [24,25], the boundary for climate change was established considering the feasibility of reducing emissions from the food system based on an emissions pathway compatible with the 2 °C climate boundary. In this scenario, it is assumed that by 2050, emissions of CO_2 from fossil fuels and land use change will need to be reduced to zero. With regard to emissions of methane (CH_4) and nitrous oxide (N_2O) from the food system, these will need to be gradually reduced, finally plateauing at approximately 4.7 Gt CO_2e in 2050. The boundary in the EAT-*Lancet* framework is therefore set at 5 Gt CO_2e (uncertainty range 4.7–5.4 Gt), including minor emissions of CO_2 from biomass burning. Hence, the boundary can be expressed as 5 Gt for CH_4, N_2O and CO_2 from biomass burning, and zero for other fossil energy-related emissions of CO_2.

Datasets on the GHG emissions associated with food sold on the Swedish market were taken from Moberg et al. [26]. These datasets account for the average emissions of CO_2, CH_4, N_2O and the hydrochlorofluorocarbon R22 (HCFC-22) directly associated with the production of the foods, following these from cradle to Swedish retail including losses and waste along the chain. The data represent the average emissions between 2011 and 2015. Minor updates were made to the datasets with updated statistics for energy use in Swedish and Dutch greenhouses [27–29] and updated fertiliser data for some crops [30]. With regard to olives used for processing to olive oil, data were adjusted to exclude drying of the olives, as this production step is only used for the production of table olives [31]. Additionally, values for extensive beef production systems with suckler cows outside Europe were adjusted to account for grazing all year around with no housing period, based on Cederberg, Meyer and Flysjö [32]. See Supplementary Material for more information.

For GHG emissions caused by the production of foods that are part of the Swedish diet, but are not included in the datasets of Moberg et al. [26], land-specific data were collected in the following order of priority based on availability: data from the World Food LCA Database [33] (available through the Ecoinvent database [34]); peer-reviewed LCA studies [35–38]; LCA reports [39–41].

All data were adjusted to match the methodology in Moberg et al. [26], i.e., to include the same factors for emissions from packaging and transportation, emissions/sequestration to and from soils due to land use, and emissions from land use change for soy-based and oil palm-based products. Additionally, all data were adjusted to account for waste and losses along the production chain, as well as to account for allocation between by-products in multi-output production systems [42–50]. Using import statistics for the largest production countries [3,30,51–54], an average value was established for each food group to represent food sold on the Swedish market.

2.2.2. Land-System Change

For land-system change, we calculated overall cropland use associated with Swedish food consumption, which is the control variable used in the EAT-*Lancet* framework [1]. The EAT-*Lancet* boundary for land-system change is based on preventing further expansion of agriculture into forest areas and other natural ecosystems. From this, a limit was established for cropland use for which a minimum cover of forest is maintained and biodiversity and key biomes are conserved at certain intactness levels.

Data for the calculations were primarily retrieved from Moberg et al. [26], using yield levels for plant-based products and feed to calculate the cropland area needed to sustain the average Swedish diet. As for GHG emissions, values for extensive beef production systems with suckler cows outside Europe were adjusted to account for grazing all year around with no housing period [32]. Data for products not included in the datasets of Moberg et al. [26] were primarily taken from the FAOSTAT statistical database [3], or otherwise taken from peer-reviewed LCA studies or LCA reports [38,39,41,47]. For honey, data were obtained from The Swedish Professional Beekeepers [55] and the European Commission [56]. All data were adjusted to match the methodology in Moberg et al. [26], using the additional data for waste, losses, allocation and import statistics, as described in Section 2.2.1.

2.2.3. Nitrogen and Phosphorus Cycling

We calculated the impacts of Swedish food consumption on the nitrogen and phosphorus cycles by using nitrogen and phosphorus application as indicators, as suggested by the EAT-*Lancet* Commission [1]. The indicator for nitrogen includes addition of 'new' reactive nitrogen to agricultural land, i.e., nitrogen from application of mineral fertiliser and from biological fixation by plants. The boundary for nitrogen is based on two aspects: limiting nitrogen concentrations in runoff water to avoid eutrophication, and maintaining a certain level of nitrogen application to feed the global population.

The control variable for phosphorus includes application of phosphorus as mineral fertiliser, for which the EAT-*Lancet* Commission set a boundary based on maximum inputs that do not lead to eutrophication of terrestrial and marine systems.

Data on mineral fertiliser application rates for nitrogen were obtained from Moberg et al. [26]. For other products not included in Moberg et al. [26], data were collected from other sources in the same order of priority as for climate change. Data on the rate of biological nitrogen fixation by plants were primarily obtained from a study by Lassaletta et al. [57]. For nitrogen fixation in pastures, data were obtained from Cederberg and Nilsson [58]. Data on phosphorus application rates were primarily obtained as site-specific data from national authorities and advisory services [59–63], and otherwise taken from the World Food LCA database [33], peer-reviewed LCA studies [46,64–66] or LCA reports [39,41,47,67].

All data were adjusted to match the methodology in Moberg et al. [26], using the additional data for waste, losses, allocation and import statistics, as stated in Section 2.2.1.

2.2.4. Freshwater Use

The use of freshwater to sustain the average Swedish diet was assessed by calculating consumptive blue water use, i.e., groundwater and surface water use in food production, which reduces the flows in watersheds by not flowing back to the same river or aquifer. This is the control variable suggested by the EAT-*Lancet* Commission [1]. The EAT-*Lancet* boundary for freshwater use is based on the estimated volume of freshwater that will be available for human use while maintaining a minimum water volume and quality to support environmental functions of river basins.

Freshwater consumption was primarily assessed for agricultural production, i.e., for irrigation of crops and for rearing of animals, as this phase accounts for the majority of the water consumed globally [1]. Inventory data on blue water consumption for the majority of food products were obtained from the WaterStat database [68,69], except for spices, for which data were retrieved from an LCA report [41]. Freshwater consumed as an ingredient in processed products such as bread and canned drinks was also accounted for. These data were primarily obtained from the Swedish National Food Agency [20]. All data were adjusted to match the methodology in Moberg et al. [26], using the additional data for waste, losses, allocation and import statistics, as stated in Section 2.2.1.

2.2.5. Biodiversity Loss

To analyse the impacts of Swedish food consumption on biodiversity loss, extinction rate was used as the control variable, as suggested by the EAT-*Lancet* Commission [1]. The boundary in the EAT-*Lancet* framework is based on limiting the rate of extinctions that will not cause irreversible changes to the Earth system.

To calculate the extinction rate from Swedish food consumption, we first estimated potential species loss (PSL) per kg of food, following the methodology in Chaudhary and Brooks [70]. Characterisation factors (CFs) were obtained for potential endemic species loss of five taxa *i* (mammals, birds, reptiles, amphibians, plants), differentiated by country *j*, for occupation of 1 m^2 of two different land use types related to food production: cropland and pasture. The CFs were then multiplied by the cropland or pasture land area in each country needed to produce 1 kg of the food (including waste, losses and allocation along the chain based on Moberg et al. [26], as stated in Section 2.2.1). From this, the overall *PSL per kg of food* in each production country *j* was calculated as:

$$PSL\ per\ kg\ of\ food_j = \sum_i (CF_{i,j,cropland} \times cropland\ area_j\ +\ CF_{i,j,pasture} \times pasture\ area_j) \quad (1)$$

To calculate the impact per kg of the average foods found on the Swedish market, the country-specific *PSL per kg of food* was multiplied by the market share of different countries for that food type:

$$PSL\ per\ kg\ of\ food\ on\ the\ Swedish\ market = \sum_j (PSL\ per\ kg\ of\ food_j \times market\ share_j). \quad (2)$$

To obtain the overall impact for the total Swedish consumption, the *PSL per kg food* was multiplied with the total consumption of different foods (i.e., total amount of food available for consumption, see Section 2.1):

$$PSL\ for\ Swedish\ \ food\ consumption = PSL\ per\ kg\ of\ food\ on\ the\ Swedish\ market \times kg\ food\ consumed \tag{3}$$

In order to benchmark the biodiversity impact to the EAT-*Lancet* boundary, expressed in extinctions per million species year (E/MSY), it was necessary to convert the *PSL*, which is the total potential species lost that will eventually take place, to a yearly extinction rate. To do this, the overall biodiversity loss (*PSL*) for the occupation of land to sustain the average Swedish diet was first allocated over a time horizon of 100 years.

To the best of our knowledge, no convention exists for the choice of such time period. However, expansion of agricultural land has escalated the last century due to an increase in human population and per capita consumption, causing severe destruction of natural habitats [71]. The choice of 100 years as a time horizon could also be argued to be in line with the choice of time period for our chosen climate metric, i.e., GWP_{100}, which is used to characterise the impacts of different GHGs (see Moberg et al. [26]). The limitations of this (arbitrary) choice of time horizon are further discussed in Section 3.4.

Finally, to obtain the E/MSY for Swedish food consumption, the PSL for Swedish food consumption per year was divided by one-millionth of the total number of recognised species (mammals [72], birds [73,74], reptiles [75], amphibians [76], plants [77]) included in the analysis.

2.3. Downscaling of the Global Boundaries

In order to benchmark the environmental impacts of the per capita Swedish diet relative to the global EAT-*Lancet* boundaries, the boundaries were downscaled to equal per capita boundaries for the global population in 2015 (7.4 billion [3]). This approach attributes equal responsibility for consumption to each global citizen and enables a straightforward and simple illustration of the contribution of the average Swede's diet to different environmental problems. Thus, this downscaling approach offers insights into how much each global citizen uses of the globally 'allowed' emissions and resource use from the food systems, regardless of where the impacts are caused. Using equal per capita boundaries also enables straightforward comparison between consumption in Sweden and other countries, regardless of population. Several other methods could be used to allocate the emissions and resource space of the global boundaries. For example, the boundaries could be allocated based on perspectives of equity of factors, such as historical emissions or resource use. Less developed countries could then be allowed higher levels of emissions or resource extraction, based on their lower contribution to the problem historically and on their ability to pay [78].

2.4. Comparisons of Global and Local Indicators and Boundaries

To identify local environmental concerns potentially not captured by the EAT-*Lancet* framework, we investigated how the SEOs relate to the EAT-*Lancet* variables and how well the global variables reflect the environmental sustainability of diets in a local context in Sweden. Based on the analysis, we sought to identify additional indicators relevant for capturing both global and local aspects in an assessment of the environmental sustainability of the Swedish diet.

The SEO framework (Figure 1) derives from *The Generation Goal*, a policy document which aims to steer Sweden's environmental policy towards solving environmental issues for the next generation without causing environmental problems outside Sweden's borders. Based on this, 16 environmental quality objectives reflecting environmental concerns of importance for the Swedish context have been established. To evaluate the progress towards achieving each objective, several indicators with different focal points are used. For example, the SEO *Reduced climate impact*, which aims at keeping the atmospheric concentration of GHGs on a level that does not threaten the climate system objective, is evaluated with four indicators, including "Atmospheric GHG concentration" and "Consumption-based emissions in Sweden and other countries" (Figure 1) [17].

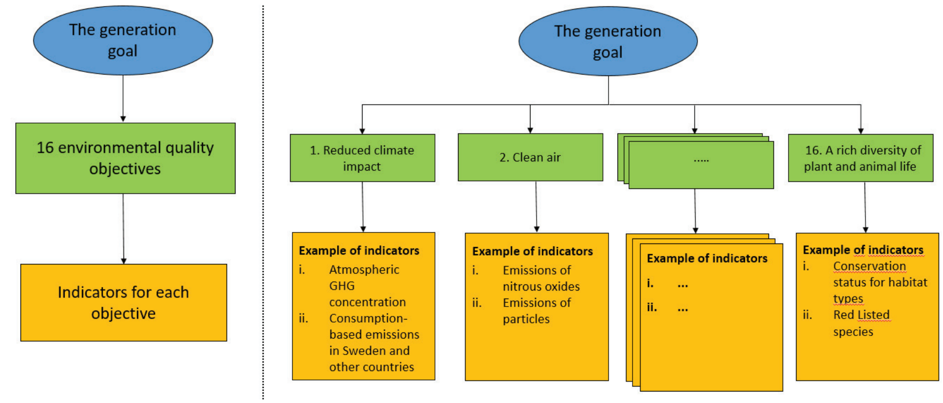

Figure 1. Illustration of (**left**) the framework for the Swedish Environmental Objectives and (**right**) examples of objectives and indicators (based on Sveriges miljömål [17]).

3. Results and Discussion

3.1. Benchmarking the Environmental Impacts of the Average Swedish Diet Relative to Global Boundaries

The environmental impacts of the average Swedish diet benchmarked relative to the EAT-*Lancet* boundaries are illustrated in Figure 2 and presented in absolute numbers in Table 2, together with per capita boundaries.

It was found that the average Swedish diet exceeded the allowed boundary for overall emissions of GHGs by more than three-fold. The boundary was transgressed with regard to emissions of CO_2, CH_4 and N_2O. Of the 2.2 ton CO_2e emitted per capita and year, emissions of CO_2 accounted for 0.92 ton (~41%), but should be zero. Emissions of CH_4 and N_2O together accounted for 1.3 ton CO_2e (~58% of total emissions), but should be below 0.68 ton CO_2e. Emissions of HCFC-22 (0.01 ton CO_2e) made up a minor fraction (<1%). Hence, even if emissions of CO_2 were reduced to zero, the boundary would still be exceeded by almost two-fold.

With regard to cropland use, the average diet required use of almost twice the cropland area per capita compared to the EAT-*Lancet* boundary. The results on GHG emissions and land use were similar to those reported by Röös et al. [14], who found that the average Swedish diet far exceeds the sustainable level of climate impact (2.5-fold the limit) and also transgresses the identified sustainable level for land use (by ~1.1-fold the limit).

Concerning application of nutrients, the Swedish diet transgressed the boundary for both nitrogen and phosphorus by more than four-fold. For consumptive water use on the other hand, the Swedish diet performed well below the boundary. For rate of extinctions, the Swedish diet caused six-fold more extinctions than the boundary. It should be emphasised that the results for extinction rate heavily depend on the choice of amortisation period for the extinctions, see discussion in Section 3.4. For all categories where the boundaries were transgressed, the impact was well above the zones of uncertainty (Table 2).

Comparison of our Swedish results against the corresponding results given for the global food consumption as assessed by the EAT-*Lancet* Commission revealed similar trends, with current (2010) global consumption exceeding the safe operating spaces for climate, phosphorus cycling and biodiversity loss while current freshwater use lay below the boundary. With regard to nitrogen cycling, the 2010 impact was above the boundary but within the range of uncertainty for the boundary. For cropland use, the global food system was still, in 2010, within the boundary but with increasing population up until 2050, the boundary was projected to be transgressed on the global level if measures to reduce waste, improve production or change diets are not imposed. However, as was seen in the results from the present study, the boundary is already exceeded for the Swedish diet.

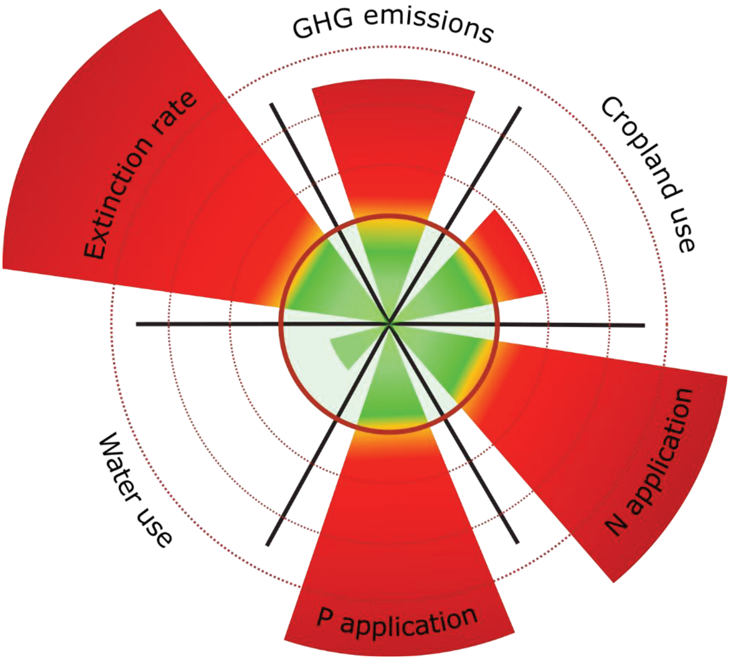

Figure 2. Environmental impacts of the average Swedish diet relative to the boundaries in the EAT-*Lancet* framework [1]. The red inner circle shows the per capita boundaries, i.e., 100% of the 'allowed' boundary, and each dotted outer circle shows transgression of the boundary by another 100%. Water use refers to consumptive water use.

Table 2. Environmental impacts of the average Swedish diet, benchmarked against downscaled per capita boundaries for the control variables given in the EAT-*Lancet* framework [1]. CO_2e = carbon dioxide equivalents, E/MSY = extinctions per million species-years. Range of uncertainty for the boundaries is given in parentheses.

Earth System Process	Climate Change	Land-System Change	Nitrogen (N) Cycling	Phosphorus (P) Cycling	Freshwater Use	Biodiversity Loss
Control variable	Greenhouse gas emissions	Cropland use	N application	P application	Consumptive water use	Extinction rate
Environmental impact per capita (results from this study)	2.2 ton CO_2e per year of which 0.92 ton CO_2, 0.82 ton CH_4 *, 0.5 ton N_2O * and 0.01 ton HCFC-22 *.	0.34 ha	57 kg N per year	5.0 kg P per year	55 m^3 per year	8.3×10^{-9} E/MSY **
Per capita boundary (downscaled from the global boundaries given by the EAT-*Lancet* Commission)	0.68 ton CO_2e per year for CH_4 and N_2O and zero for CO_2 from fossil fuels and land use and land use change (0.68–0.73)	0.18 ha (0.15–0.2)	12 kg N per year (8.8–18)	1.1 kg P per year (0.8–2.2)	339 m^3 per year (136–542)	1.4×10^{-9} E/MSY (1.4×10^{-10}–1.1×10^{-8})

* Expressed in CO_2e. ** Allocated over 100 years (see Section 2.2.5).

3.2. Relative Contribution of Foods to Environmental Impacts of the Average Swedish Diet

Figure 3 provides an illustration of the environmental impacts broken down per kg of food on the Swedish market and Figure 4 presents the environmental impacts per capita from the overall diet and the relative contribution from different food groups to each impact category. For more detailed results, see Supplementary Material (Tables S2 and S3).

Looking at larger food categories, animal products contributed the largest share of GHG emissions (about 67%), 18% were caused by the consumption of sweets, snacks and drinks (excluding milk) and the remaining 15% were caused by the consumption of other plant-based foods (Figure 4). A similar trend was seen for cropland use per capita and use of nitrogen, with animal products causing the largest impact (60% and 77% respectively). With respect to phosphorus application, animal products contributed 38% of the overall impact. The consumption of sweets, snacks and drinks contributed between 12% and 42% of the overall impacts for the mentioned categories, with the lowest contribution for nitrogen application and the highest for phosphorus. Other plant-based products contributed between 10% and 19% of the overall impact for these categories (lowest for application of nitrogen and highest for phosphorus). Sweets, snacks and drinks as a group, thus made the highest contribution to phosphorus application. This group also made the highest contribution to species extinction rates with 45% of the overall impact whereas other plant-based products and animal products contributed 26% and 27% of the overall impact respectively. Finally, for consumptive water use, the contribution was highest from plant-based products with 48% of the overall impact. Animal products and sweets, snacks and drinks caused similar impacts with 28% and 24% of the overall impact respectively.

The low contribution from many plant-based foods to the environmental impacts of GHG emissions and cropland use (Figure 4) is mainly explained by the relatively low impact per kg for products such as fruits, leafy vegetables, root vegetables and cereals (Figure 3), which is in line with earlier findings [7,48]. Important exceptions with regard to GHG emissions and cropland use per kg were found, e.g., for coffee, cocoa and vegetable oils (especially olive oil), for which cropland use made an important relative contribution to the overall impact (Figure 4).

Sweets, snacks and drinks and the category of other plant-based foods contributed relatively more than animal products to the categories of extinction rate, consumptive water use and phosphorus application (Figure 4). For biodiversity, this was explained by the high impact per kg of food caused by plant-based products such as vegetable oils (especially olive oil), fruits, nuts, coffee, cocoa and rice (Figure 3). Together with high consumption of these foods, this led to important overall impacts (Figure 4). The high biodiversity impact per kg of olive oil, coffee and cocoa was mainly explained by the high cropland use, while for products such as bananas, which are imported from South and Central America, the occupation of land for production in these areas caused high impacts due to high biodiversity loss per occupied m^2. In general, animal products such as beef caused low biodiversity impacts per kg despite high land use (Figure 3), due to that most livestock production for the Swedish market take place on relatively biodiversity-poor land (Sweden and Northern Europe). However, the impacts on biodiversity loss would change considerably if production were to take place in countries where the occupation of land causes higher biodiversity loss per occupied m^2 in comparison to countries that currently represent the largest shares on the Swedish market, such as Sweden, Ireland, Poland and Germany. An important exception was seen for lamb, which was found to have the highest biodiversity impacts per kg (Figure 3) and also gave a high contribution to the overall impact (Figure 4), despite low consumption rates. This was explained by its high land use (especially pasture), together with the high biodiversity loss from occupation of land for sheep production in New Zealand, a country which represents about 20% of the Swedish market (Supplementary Material).

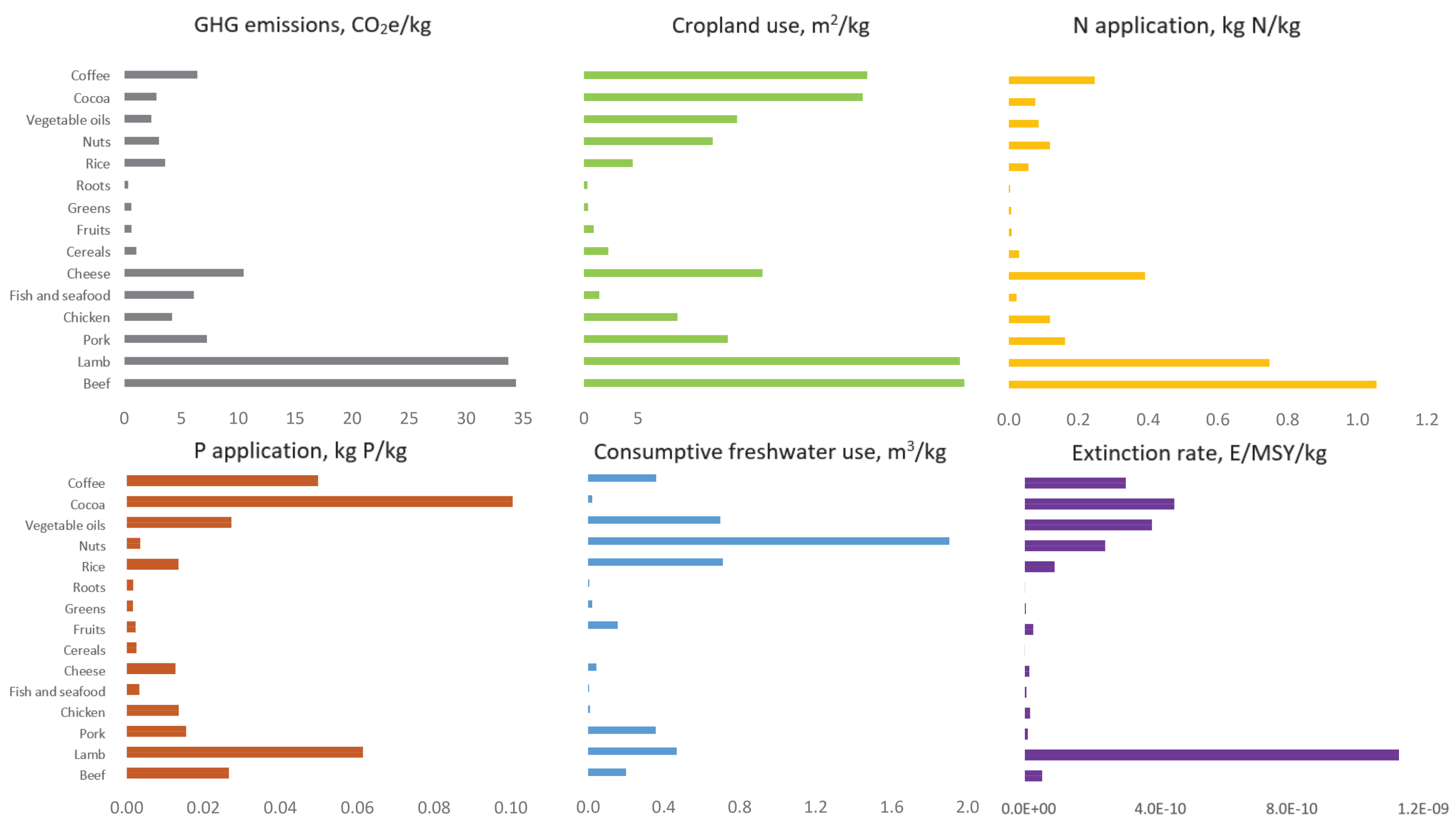

Figure 3. Environmental impacts per kg of food (per kg bone-free weight for meat and edible weight for fish and seafood) for food products on the Swedish market.

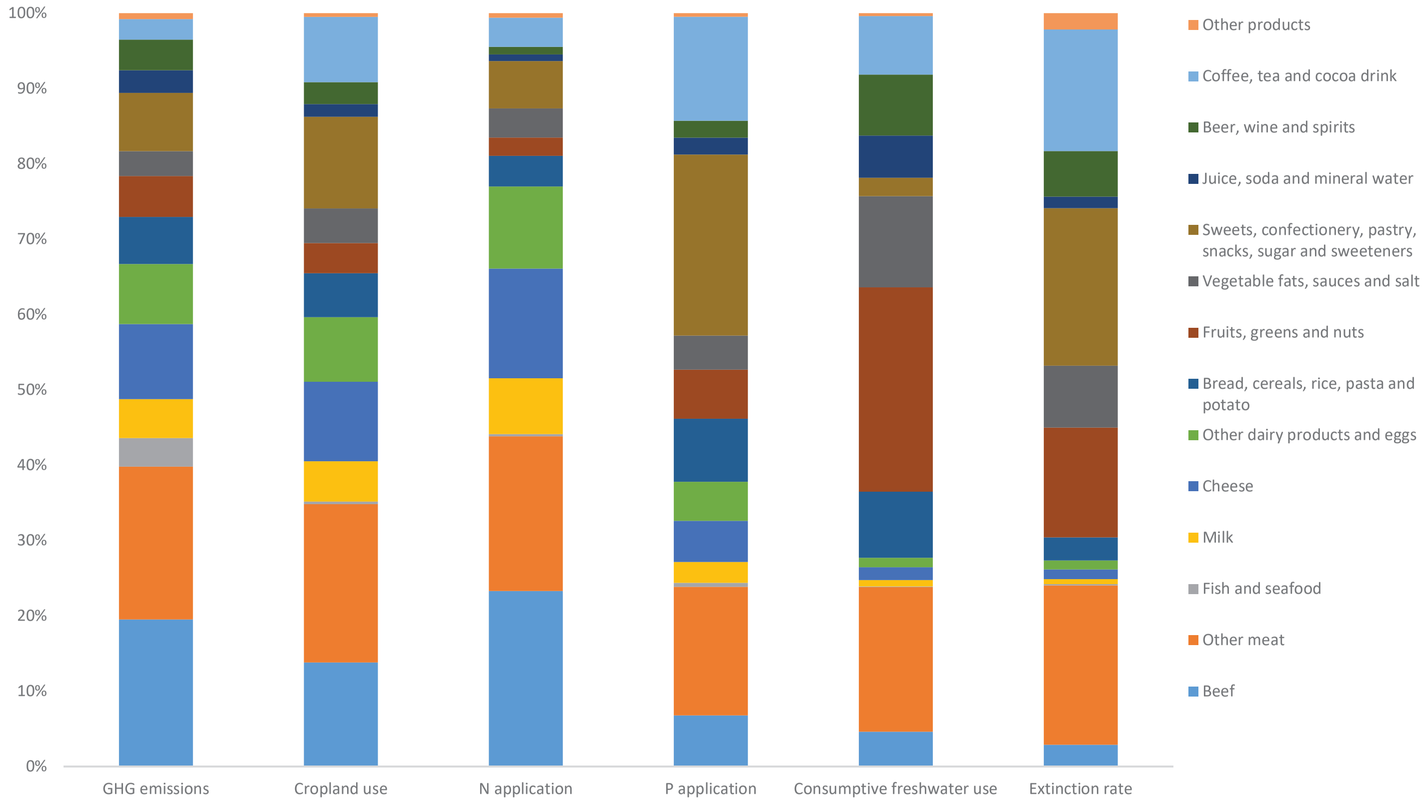

Figure 4. Relative contribution by different food groups to the environmental impacts from Swedish food consumption.

Freshwater consumption per kg was especially high for nuts (almonds in particular), rice and vegetable oils (Figure 3), as also seen in its relative contribution (Figure 4). High freshwater consumption per kg was also seen for coffee and fruits in comparison with other plant-based products (Figure 3). An important share of the relative contribution was made by fruits and leafy vegetables (Figure 4), because, e.g., a large proportion of fruits are imported from areas where high irrigation levels are often required (see Supplementary Material). With regard to Swedish products, irrigation is generally carried out on a small proportion of Swedish agricultural land, with crops that often require irrigation including root crops, vegetables and fruits [79]. Grains and ley for animal feed and pasture are seldom irrigated, either in Sweden or in other production countries [69,79]. Due to freshwater use for rearing animals [68], animal-based products still had higher consumptive freshwater use per kg than many plant-based products (Figure 3).

With regard to phosphorus application, fertiliser application was generally low per kg for plant-based products except for cocoa, coffee and olive oil (Figure 3), for which the application rates were found to be particularly high. This was also reflected in their relative contribution (Figure 4).

The highest impact on the climate and several other environmental categories per kg of food was found for ruminant meat, i.e., beef and lamb (Figure 3), which is in line with earlier findings [7,48]. This led to important contributions to the impacts of the average diet for all variables, but were especially pronounced for GHG emissions, cropland use and nitrogen application (Figure 4). Pork, chicken, processed meat products and dairy products such as cheese also had high environmental impacts per kg (Figure 3) and made a high contribution to all impacts (Figure 4). The impacts of fish and seafood varied depending on fish species (Supplementary Material), but the relative contribution to the overall impacts was generally low compared with that of other animal products (Figure 4).

3.3. Comparison of Global and Local Indicators and Boundaries

Clear links were identified between the scope of several of the SEOs and the aim of one or more of the boundaries of the Earth system processes in the EAT-*Lancet* framework (Figure 5). Below, we discuss how the SEOs link to each of the Earth system processes in the EAT-*Lancet* framework and assess how well the global boundaries reflect environmental sustainability in the local context (Sections 3.3.1–3.3.6). Based on this, we suggest additional indicators that capture both global and local aspects in assessment of the environmental sustainability of the average Swedish diet (Section 3.3.7).

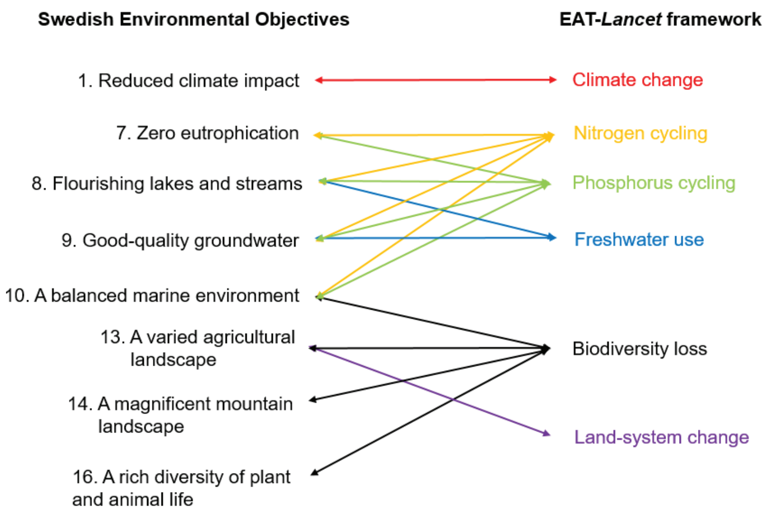

Figure 5. Links between Swedish Environmental Objectives [17] and the EAT-*Lancet* framework [1].

3.3.1. Climate Change

With regard to *Reduced climate impact* (SEO 1), the scope of the SEO is clearly in line with the aim of the boundary on *climate change* in the EAT-*Lancet* framework, as both are based on limiting global warming to a maximum of 2 °C above pre-industrial levels [1,17]. The scope of the boundary in the EAT-*Lancet* framework has a global perspective and matches one of the indicators used for assessing the SEO, i.e., "Consumption-based emissions in Sweden and other countries". This indicator measures emissions from Swedish consumption, caused in Sweden and other countries. For this indicator, food-related emissions are reported separately. However, as no specific boundary has been set for food-related emissions for the SEO indicator, it does not permit assessment of the absolute environmental sustainability of the Swedish diet. Such assessment is possible using the EAT-*Lancet* boundary.

3.3.2. Land-System Change

The scope of *A varied agricultural landscape* (SEO 13) relates to the Earth system process of *Land-system change* in the EAT-*Lancet* framework, as both aim at preserving biodiversity and key biomes [1,17]. Interestingly, while the EAT-*Lancet* boundary was set to limit further expansion of agricultural land globally, the SEO aims at maintaining current Swedish agricultural land. The rationale for the SEO is the decline in agricultural land in Sweden in recent decades, caused by reduced profitability and rationalisation of agricultural holdings, fewer grazing animals and expansion of settlements and infrastructure on agricultural land, which threatens farmland-associated biodiversity [80]. While the benchmarking of the Swedish diet against the EAT-*Lancet* boundary serves to emphasise the need for a more land-efficient diet, the SEO offers the insight that agricultural production in Sweden could be maintained or even increase. Considering cropland availability within the country, Sweden could thus become a net exporter of food, i.e., increase food production for both domestic and international markets. The SEO *A varied agricultural landscape* also includes an indicator for the preservation of pastures, in order to protect important biodiversity-rich biomes such as semi-natural pastures by continued grazing by animals. As the EAT-*Lancet* focuses on cropland use, this is an important addition to capture local environmental sustainability aspects in Sweden. The SEO also captures quality aspects of land use that are important for productivity, such as soil fertility and drainage, which could be an important complement to the EAT-*Lancet* framework, although this aspect might fall into the category of social sustainability rather than environmental sustainability.

3.3.3. Nitrogen and Phosphorus Cycling

The SEOs on *Zero eutrophication* (SEO 7) and *A balanced marine environment, flourishing coastal areas and archipelagos* (SEO 10) are partly captured by the EAT-*Lancet* indicators of *Nitrogen cycling* and *Phosphorus cycling*, as these focus on limiting emissions of nutrients in order to reduce eutrophication of terrestrial and marine ecosystems [17]. The EAT-*Lancet* boundaries are related to the amount of new reactive nitrogen and phosphorus added [17] to ecosystems on a global level, i.e., nitrogen from synthetic fertiliser or via fixation in legume crops and mined phosphorus [1]. While 'added nutrients' can serve as a proxy of the risk of eutrophication to sustain Swedish food consumption, they do not consider local aspects, including the status of the aquatic recipient of the nutrients. In contrast, the SEOs focus specifically on limiting emissions of nutrients to local recipients such as the Baltic Sea and on the eutrophication status of the recipients. The SEOs thus directly target regional problems of eutrophication, which gives a complementary view of impacts. However, current indicators in the SEOs are designed to measure overall emission loads from Sweden and neighbouring countries and are difficult to link to specific foods and diets. Furthermore, although boundaries have been set which serve to benchmark Sweden's overall territorial performance, no specific limit has been set for the maximum emissions load from specific sectors such as agriculture. Hence, it cannot be used to benchmark impacts from the diet. Moreover, as the indicators are production-based, they do not cover eutrophication impacts caused in other parts of the world due to Swedish consumption of food. Site-dependent eutrophication models can be used for such assessments, which would also enable estimation of the impacts on local recipients (e.g., [81,82]). However, these models require detailed data on, e.g., emission intensities for specific catchments. Due to limited availability of emissions data and limited traceability of foods to the Swedish market at that level of detail, it is currently very difficult to carry out such an assessment of the whole Swedish diet.

3.3.4. Freshwater Use

The SEOs *Flourishing lakes and streams* (SEO 8) and *Good-quality groundwater* (SEO 9) include aspects that are partly captured by the EAT-*Lancet* Earth system process of *Freshwater use*. However, while the latter addresses both surface and groundwater, these water sources are considered separately in the two SEOs. Moreover, the SEOs address both the quantity and quality of water sources in a regional perspective, while the EAT-*Lancet* boundary only focuses on quantity of water used from a global viewpoint, i.e., more from the perspective of water as a resource [1,17]. Quality aspects of water pollution are instead, to a certain extent, covered by the EAT-*Lancet* variables for nitrogen and phosphorus application. Hence, there is a link between the SEOs *Flourishing lakes and streams* and *Good-quality groundwater* and the Earth system processes *Nitrogen cycling* and *Phosphorus cycling* (Figure 5). When applying a global perspective on freshwater use, there is a risk of overlooking regional variations in water scarcity. Therefore, an indicator for freshwater use should ideally include aspects of local water availability [16]. The SEOs offer this on a national level for Sweden, but the methodology is currently limited to analysing the overall status of water scarcity of, e.g., groundwater resources, rather than impacts from a diet perspective. Impacts of local scarcity could instead be assessed in a consumption-based analysis where water consumption in a certain area is weighted according to the local availability [83,84]. This would require more detailed inventory data, ideally on catchment level.

3.3.5. Biodiversity Loss

Apart from focusing on limiting excessive nutrient application, the SEO *A balanced marine environment, flourishing coastal areas and archipelagos* (SEO 10), also has a link to the EAT-*Lancet* Earth system process of *Biodiversity loss*, as both aim at conserving marine and terrestrial biodiversity [1,17]. With regard to marine biodiversity, today, about 90% of global fish stocks are estimated to be overfished or fished at capacity [1]. Due to methodological limitations, however, it may be difficult to account for the extinctions of marine species on a global level. On a national level, the SEO includes an indicator that focuses specifically on sustainable fish stocks in Swedish fishing waters, which could be used as a complement to the EAT-*Lancet* boundary. However, as 75% of the fish and seafood in the average Swedish diet is imported [85], an additional indicator of the status of fish stocks should ideally be used to include the international perspective.

Other SEOs also have clear links to the Earth system process *Biodiversity loss* in the EAT-*Lancet* framework, including *A varied agricultural landscape* (SEO 13), *A magnificent mountain landscape* (SEO 14) and *A rich diversity of plant and animal life* (SEO 16). While *A rich diversity of plant and animal life* focuses on biodiversity conservation in general, *A varied agricultural landscape* and *A magnificent mountain landscape* concentrate on conservation of biodiversity in agricultural land and mountain areas, respectively [17]. The methodology used in this paper allows for site-specific evaluation of biodiversity impacts from land use [70,86]. However, the EAT-*Lancet* boundary for benchmarking the environmental sustainability of biodiversity impacts is set on a global level, with an overall threshold for species extinctions. As biodiversity impacts are primarily manifested on a local or regional level, the SEOs targeting biodiversity impacts can therefore serve as important complements to the global EAT-*Lancet* boundary. Regarding *A magnificent mountain landscape*, one of the indicators used in the assessment considers reindeer grazing, as this is a prerequisite for conservation of threatened species in mountain areas of Sweden [17]. However, the indicator does not state a certain threshold for the area or number of animals needed for landscape maintenance, and thus, it is currently difficult to assess this aspect. One of the indicators for *A rich diversity of plant and animal life* is represented by an index of the state of a threatened species in certain areas in Sweden [17]. Agricultural land is one such area and the indicator could therefore be used to assess local aspects of biodiversity impacts. Similarly, one of the indicators for the SEO *A varied agricultural landscape* is an index focusing on population trends in birds and butterflies on farmland [17]. However, it is difficult to link the impacts measured with these indices to a diet, and the indicators can currently be used only to assess overall territorial performance of Swedish agriculture.

3.3.6. Additional Aspects not Captured by the EAT-*Lancet* Framework

Several of the SEOs were found to have no direct link to the Earth system processes in the EAT-*Lancet* framework. Many are part of the Planetary Boundaries framework [24,25], but, to the best of our knowledge, no boundary related to the food system has been set for these. Nevertheless, they provide important aspects for assessing the environmental sustainability of the Swedish diet. For example, *Clean air* (SEO 2) relates to environmental impacts of diets due to emissions of nitrogen oxides (NO_x) and particles, where agricultural activities contribute 12% and 10%, respectively, of overall emissions in Sweden [87]. Indicators including these pollutants could therefore serve to capture local environmental sustainability issues. Furthermore, it may be relevant to include aspects of *Natural acidification only* (SEO 3) in environmental assessments of the Swedish diet, as a large proportion of acidifying emissions in Swedish production originates from agriculture. A key issue within agriculture is ammonia emissions from manure management [88], which could therefore be used as an indicator.

Food production is highly relevant for the SEO *A non-toxic environment* (SEO 4) due to the use of pesticides in agriculture, which can cause damage to humans, animals and ecosystems [17]. This could be included in an environmental assessment of diets through an indicator on the use of pesticides. As the majority of the current Swedish pesticide footprint arises in production abroad, it is important to assess pesticide use in both domestic production and production of imported products, as suggested by Steinbach et al. [89]. However, there is currently limited availability of data on pesticide use for countries outside Europe [89], so better data are needed to increase the accuracy of such assessments.

A protective ozone layer (SEO 5) links to food production by emissions of chlorofluorocarbons (CFCs) and nitrous oxide (N_2O). In food production, CFCs arise due to the use of refrigerants in the fishing industry, but these are currently being phased out under an EU directive [90]. This affects the largest fishing countries that export fish to Sweden [26]. The main concern with regard to impacts on ozone depletion from the food system is, instead, N_2O emissions. These emissions, which mainly arise due to fertilisation of arable land and manure management, currently have greater depletion potential than any other ozone-depleting gas [91]. Including emissions of N_2O as an indicator of stratospheric ozone depletion in assessment of environmentally sustainable diets is therefore important.

The SEOs where no direct connection was found relative to diets and which were not considered relevant for further analysis are presented in the Supplementary Material (Table S1).

3.3.7. Summary of Comparisons of Global and Local Frameworks and Suggested Indicators

Based on the discussion in Sections 3.3.1–3.3.6, Table 3 summarises aspects in the SEO framework that are not covered by the EAT-*Lancet* framework and lists areas where additional indicators could be developed for assessing these aspects. The indicators for assessing the environmental impacts of the Swedish diet (Table 3) should be consumption-based, i.e., cover impacts both from food produced within Sweden and from imported food. For this, there is a need for better availability of inventory data on resource use and emission intensities on a detailed level and for consideration of environmental aspects in import countries that might not be an issue in Sweden. There is also a need for better traceability of foods entering the Swedish market. Furthermore, indicators should ideally be designed together with specific environmental boundaries, to enable benchmarking at the level of detail for foods or diets.

Table 3. Summary of comparisons between global and local indicators, aspects not covered by the EAT-*Lancet* framework, suggested indicators and areas where new data or methods are needed. SEO = Swedish Environmental Objectives, GHG = greenhouse gases, NOx = nitrogen oxides.

Earth System Process	Aspects in the SEO not Covered by the EAT-*Lancet* Framework and Other Aspects Covering Environmental Sustainability	Suggested Indicator	Need for Additional Data or Method Development
Climate change	-	GHG emissions	-
Land-system change	Maintain Swedish agricultural land, quality aspects of land use	Swedish agricultural land and soil fertility aspects	System for monitoring soil fertility that can be connected to foods. Soil organic content could potentially be used.
	Maintain Swedish pasture, including semi-natural pastures	Pasture use	Improved statistics on different land types and uses of pasture
Nitrogen and phosphorus cycling	Site-dependent eutrophication impacts, i.e., emissions to specific catchments and nutrient status of recipients	Site-dependent eutrophication impacts	Data on emission intensities for specific catchments and on nutrient status of recipients
Freshwater use	Site-dependent impacts of consumptive freshwater use	Site-dependent consumptive freshwater impacts	Data on consumptive freshwater use and availability on catchment level
Biodiversity loss	Local aspects of biodiversity, e.g., state of threatened species on agricultural land	Terrestrial extinction rate	Methods that allow local impacts to be linked to foods
	Marine extinction	Marine extinction rate	Methods that allow local impacts to be linked to foods
Atmospheric aerosols	Air pollution	Emissions of NOx and particles	-
Acidification of freshwater and land	Acidification of freshwater and land	Emissions of ammonia	-
Chemical pollution	Toxic substances to the environment	Pesticide use	Data on the type and amount of pesticides used for different crops, especially for outside the European Union. Methods like UseTox [92] can then be applied.
Ozone depletion	Emissions of ozone depleting substances	Emissions of N_2O	-

Another important framework for sustainability assessment is, naturally, the United Nations Sustainability Development Goals (SDGs) [93]. Ridoutt, Hendrie and Noakes [16] investigated the extent to which the current literature on sustainable diets covers the aspects included in the SDG targets. They found 14 different environmental areas of concern in the SDGs: Water scarcity, Natural resource depletion, Urban air quality, Ozone depletion, Human and ecotoxicity, climate change, Marine debris, Marine eutrophication, Freshwater ecosystem quality, Depletion of fish stocks, Deforestation, Land degradation and desertification, Biodiversity loss and Invasive species. All, except Marine debris and Invasive species, are covered in our discussion on the SEO in relation to the EAT-*Lancet* global boundaries. Developing indicators to relate these two missing aspects to diets is probably challenging, as it is difficult to relate the amount of marine debris and invasive species to specific foods, and hence, diets.

3.4. Study Limitations

The global boundaries, indicators and corresponding inventory data used to assess the environmental impacts of Swedish food consumption in this study are all associated with uncertainties and limitations and thus, there is potential for increasing the accuracy of the results in future research.

As for setting absolute global boundaries for the food system, as highlighted by the EAT-*Lancet* authors, this is highly challenging since the drivers of Earth system processes are complex and interconnected. In addition, some of the EAT-*Lancet* boundaries have been criticised for not relating to the original absolute threshold levels of the Planetary Boundaries, i.e., based on absolute biophysical limits for Earth systems within which humanity should operate. The boundaries for GHG emissions and nitrogen application are, instead, based on the unavoidable share of emissions and resources needed to feed the global population. Einarsson, McCrory and Persson [94] pointed out that in order for the boundaries to be scientifically consistent, they should rely upon scientific evidence on the limits of the Earth systems, although this causes trade-offs between reaching environmental targets and maintaining current levels of prosperity.

As for calculating the environmental impacts from the Swedish diet for different indicators, these assessments are also associated with model and data uncertainties. For calculating cropland use, there is, in general, good data availability on yield levels through statistics databases (e.g., [3,18]). Further, as the indicator focuses solely on one variable, i.e., crop productivity levels, calculations are straight-forward. For GHG emissions, on the other hand, important emissions arise in several process steps in the life cycle of various food products. In many of these steps, emissions are variable due to, e.g., climate conditions and soil characteristics. Furthermore, different methodological choices can be made to account for the emissions, which can substantially affect the results, e.g., when accounting for emissions from land use and land use change. Other limitations to assessment of GHG emissions include lack of detailed inventory data for countries outside Europe and lack of data on food groups such as fish and seafood [26]. For example, the GHG emissions for meat on the Swedish market have been found to vary from approximately −40% to +100% [95]. Uncertainties are always important to consider, and even more so when benchmarking against absolute boundaries. Establishing uncertainty ranges for the environmental impacts of the Swedish diet is, hence, an important topic for coming studies, but is associated with major difficulties due to data limitations, e.g., on variations in input data, that become increasingly important as impacts are reduced to fit within the boundaries.

With respect to nitrogen and phosphorus application, site-specific data from statistical databases or advisory services are primarily available for Sweden and other European countries (e.g., [30]), while data for production countries outside Europe mainly are available through databases (e.g., the World Food LCA Database [33]), peer-reviewed studies or LCA reports.

Regarding consumptive freshwater use, inventory data for the present study were primarily obtained from the WaterStat database [68,69]. A limitation in the inventory data is that consumptive water use for crops does not necessarily represent the actual water consumed. Rather, it is based on modelling crop water requirements using inventory data on crop parameters and climate parameters such as temperature and precipitation [69].

Concerning estimation of potential extinctions due to land occupation, there are several uncertainties, deriving from both general modelling and variables, and from data gaps and uncertainties in inventory data, in the methodology developed by Chaudhary and Brooks [70]. There is potential to extend the modelling to include additional land use classes (e.g., by distinguishing between annual and permanent crops) and taxa (e.g., by including invertebrates) [70]. For these indicators, data on uncertainties are largely missing; a gap that needs to be filled in future research. Furthermore, the choice of time horizon for allocation of overall potential species loss had to be chosen arbitrarily, which had large impacts on the results for biodiversity loss. For example, allocating all of the impacts to the same year would, naturally, lead to a 100 times larger impact, which would be 600-fold the EAT-*Lancet* boundary. Using 20 years would show impacts 30 times the boundary while allocating the species loss over 500 years would cause impacts 1.2-fold the boundary.

Another limitation in the present study relates to the food supply data, which were obtained from the statistical database of the Swedish Board of Agriculture [18]. For some of the product groups, e.g., vegetable fats, sauces, fish and seafood, detailed statistics are lacking and assumptions have to be made based on, e.g., food surveys and reports [87,96].

4. Conclusions

The environmental impacts of the average Swedish diet were shown to exceed the global EAT-*Lancet* environmental boundaries for GHG emissions, cropland use and application of nitrogen and phosphorus by two- to more than four-fold. For extinction rate, the boundary was exceeded by nearly six-fold. The only environmental category for which the global boundary was not transgressed was freshwater use, where the impact of the diet was well below the limit.

Comparisons of global and local indicators for assessing the environmental sustainability of Swedish food consumption revealed that the EAT-*Lancet* variables cover many aspects included in the SEOs, such as reducing emissions of GHGs and limiting input of nutrients to ecosystems in order to reduce eutrophication of terrestrial and marine ecosystems. While these global indicators capture the overall impact of diets from a 'global allowance' perspective, for many aspects, more fine-resolution indicators are needed to capture actual impacts in the local context. For example, when assessing eutrophication impacts, site-dependent variables should ideally be included, e.g., emissions intensities to specific catchments and nutrient status of recipients.

Aspects in the SEOs not covered by the EAT-*Lancet* variables include chemical pollution and acidification of freshwater and land. Such aspects could be covered by additional indicators, but absolute boundaries for these are currently lacking.

To enable inclusion of complementary aspects covering the environmental sustainability of diets, there is a need for reliable inventory data on resource use (e.g., for pesticide use) and emission intensities on a detailed level (e.g., for nutrients to assess eutrophication impacts), together with better traceability data for foods imported to the Swedish market.

Supplementary Materials: The following are available online at http://www.mdpi.com/2071-1050/12/4/1407/s1, Table S1: Scope of SEOs where no direct connection was found relative to diets and which were not considered relevant for further analysis, Table S2: Environmental impacts of food products and categories in the Swedish diet, per kg of food and per capita, Table S3: Benchmarking of the Swedish diet relative to the EAT-Lancet boundaries with % of performance relative to each boundary together with lower and higher uncertainty boundaries, Table S4: Overview of modeling choices and food supply data, Inventory data.

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IV

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Taxing food consumption to reduce environmental impacts – Identification of synergies and goal conflicts

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ABSTRACT

This study analysed the environmental impacts of taxation on Swedish food consumption and sought to identify potential synergies and goal conflicts between environmental aspects. This was done by analysing various taxation scenarios to reduce environmental impacts of food, including taxation based on: climate impact; a score based on weighting of several environmental impacts; and adjusted rates of value-added tax (VAT).

A net decrease in food consumption was seen for most taxation scenarios, resulting in reduced burdens for climate change and most other environmental categories. An exception was found for a scenario simulating reduced VAT rates for plant-based products, where a net increase of food consumption was seen, resulting in an increased burden for all environmental categories. Many of the scenarios resulted in a decrease in beef consumption, and hence a decline in pasture use. This is positive from a global perspective by limiting expansion of agricultural land, but on regional level in Sweden it could cause a goal conflict with maintaining biodiversity-rich semi-natural pastures. To avoid this, beef production on semi-natural pastures could be further incentivised by production-side measures. With regard to biodiversity loss, the overall burden could increase if taxation leads to an increase in products from biodiverse regions.

1. Introduction

The food system is estimated to cause one-third of global anthropogenic greenhouse gas (GHG) emissions, and has therefore been identified as a major driver of climate change (Crippa et al., 2021). Transformation of the food system is urgently needed, calling for both production- and consumption-side measures, where the latter involve e.g. reducing food waste and over-consumption of food, as well as changes in dietary patterns (IPCC, 2019, Willett et al., 2019). Studies have found that animal products have a substantially larger climate impact than most plant-based foods, indicating potential for reducing environmental burdens by shifting from diets with a large share of animal-based foods towards diets with more plant-based foods (e.g. Hallström et al., 2015, Poore and Nemecek, 2018, Willett et al., 2019). However, previous research suggests that changing people's dietary patterns can be challenging (Hartmann and Siegrist, 2017) and has limited potential to be driven by voluntary actions (Garnett et al., 2015). Policy instruments have therefore been identified as necessary and, in particular, a climate tax on food consumption has been suggested in countries such as Sweden (Lööv et al., 2013, SSNC, 2015), Germany (TAPPC, 2020a), and the Netherlands (TAPPC, 2020b). The effects of a climate tax on food have also been modelled in the literature, where e.g. Springmann et al. (2016) found potential for a 9% reduction in food-related GHGs when analysing global pricing of food based on the climate impact. On regional level, Säll et al. (2020) found that a climate tax targeting food on the Swedish market could decrease food-related GHG emissions by up to 10%. Other proposals for changing food prices in order to steer consumption in a desired direction include changes to the existing value-added tax (VAT) system, e.g. increasing the tax rate on foods with a high climate impact or subsidising foods with a low impact (Ekvall et al., 2016, Broeks et al., 2020).

Previous modelling studies on taxing food have primarily focused on the effects on climate impact following the introduction of a tax. However, a shift in dietary patterns due to food taxation would also affect other environmental impacts caused by food production. Apart from generating GHG emissions, food production causes environmental pressures such as appropriation of land and freshwater resources. The use of fertilisers and chemical substances in agriculture leads to pollution of land and water, and poses risks to human health and biodiversity (Willett et al., 2019). Studies investigating the environmental impacts of

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food and the implications of changed dietary patterns have found synergies between many of these environmental impacts, which can be explained by the common underlying drivers of environmental damage (Röös et al., 2013, Aleksandrowicz et al., 2016, Martin and Danielsson, 2016, Martin and Brandão, 2017, Chaudhary et al., 2018, Poore and Nemecek, 2018).

A study by Aleksandrowicz et al. (2016) found that consumption changes towards diets with a larger share of plant-based foods have the potential to reduce GHG emissions and land use by 70%, as well as reducing water use by 50%. Martin and Brandão (2017) investigated the environmental implications of Swedish dietary choices and concluded that replacing meat with more plant-based foods in diets could reduce climate impact, eutrophication, acidification and land use, and be beneficial for biodiversity. However, those authors also pointed out that a shift towards more plant-based diets could alter the effects of toxicity on humans and terrestrial ecosystems, due to higher prevalence of metals in e.g. cereals and oilseed crops. Such possible goal conflicts between environmental categories due to changes in dietary patterns were also highlighted by Martin and Danielsson (2016), who found that if animal-based protein sources such as beef and pork were to be replaced by larger shares of poultry, water consumption would increase. A study by Nordborg et al. (2017) found that pork and chicken, which generally have lower climate impact than beef, may have a higher impact on freshwater ecotoxicity than beef. Moreover, reducing consumption of climate-burdening food such as beef could lead to a potential goal conflict between climate mitigation and biodiversity conservation in Swedish pastures, as grazing animals are important to maintain semi-natural pastures and threatened species within these (e.g. Lööv et al., 2013).

More knowledge is needed on how taxation to reduce environmental impacts affects environmental outcomes more broadly than only climate impact. Analyses of synergies between environmental categories and the potential for simultaneous reductions in environmental burdens are important in the design of efficient policy instruments. Likewise, identification of potential goal conflicts is important to avoid implementing policy instruments which may have negative effects in other areas. The aim of this study was therefore to investigate the environmental impacts of different taxation scenarios on Swedish food consumption, in order to identify potential synergies and risks of goal conflicts between environmental aspects.

2. Material and methods

Different food taxation scenarios that resulted in different price changes for foods were developed (described in Section 2.1). The effect on food consumption arising from these changes in food prices was estimated using a demand system in which historical price and consumption data were employed to estimate price elasticities for foods on the Swedish market. The demand system is fully described in Säll et al. (2020) and briefly outlined in Section 2.2. The changes in consumption were then used to estimate the energy intake of the Swedish population (described in Section 2.3) and the effects on the environmental impact resulting from the different taxation scenarios, using data on the environmental impacts of foods on the Swedish market (described in Section 2.4).

2.1. Taxation scenarios

The taxation scenarios are summarised in Table 1. The first set of scenarios considered were all based on taxing the GHG emissions caused by production using a tax rate of 1.15 SEK per kg GHG, which corresponds to the 2015 Swedish tax on CO_2 emissions (Swedish Tax Agency, n.d.-a). Applying the same tax for all emissions sources (here food and energy) ensures that emission reductions are cost-efficient, i.e. reductions will be made where they are cheapest (Baumol and Oates, 1988). However, the scenarios differed with regard to the products subjected to taxation (described in Section 2.1.1), the system boundaries used in calculation of emission intensity of the foods (Section 2.1.2), and the weighting of emissions of different GHGs (Section 2.1.3). In an additional scenario, taxation based on weighting of several environmental impacts was simulated (Section 2.1.4). However, the chosen tax levels were based on the cost initially used for GHG emissions, and were thus not linked to the related environmental damage costs. There are few, and possibly outdated, studies investigating the costs of nutrient leaching per kg emissions abatement for the Baltic Sea. It is also difficult to link costs of nutrient abatement and biodiversity loss to individual

Table 1
Scenarios used to simulate consumer responses to taxation targeting the environmental impacts of food consumption. GWP_{100} = Global Warming Potential over 100 years; GTP = Global Temperature Potential, see Section 2.1.3.

	Taxation based on different sets of food products	Taxation based on different system boundaries	Taxation based on different weighting of GHG emissions	Taxation based on weighting of several environmental impacts	Adjustment of VAT rates
Products included in scenarios	i. All products ii. Animal products iii. Beef iv. Monogastric meat and eggs	i-iii. All products	i-iv. All products	i. All products	i. Animal products ii. Animal products; fruit, vegetables, cereals[1] iii. Fruit, vegetables, cereals[1]
System boundary in scenarios	i-iv. To retail-gate	i. To retail-gate ii. To farm-gate iii. Agriculture	i-iv. To retail-gate	i. To retail-gate	–
Weighting of impacts in scenarios	i-iv. GHG emissions weighted with GWP_{100} with climate-carbon feedbacks	i-iii. GHG emissions weighted with GWP_{100} with climate-carbon feedbacks	i. GHG emissions weighted with GWP_{100} with climate-carbon feedbacks ii. GHG emissions weighted with GWP_{100} without climate-carbon feedbacks iii. GHG emissions weighted with GTP_{100} with climate-carbon feedbacks iv. GHG emissions weighted with GTP to limit warming to 2 °C	i. Environmental impacts weighted according to 'distance to target'	–
VAT rate in scenarios	i-iv. No change, 12% on all products	i-iii. No change, 12% on all products	i-iv. No change, 12% on all products	i. No change, 12% on all products	i. Increased VAT rate on animal products to 25% ii. Increased VAT rate on animal products to 25%; reduced VAT rate on fruit, vegetables and cereals[1] to 6% iii. Reduced VAT rate on fruit, vegetables and cereals[1] to 6%

[1] This includes fruit, root vegetables, brassicas, onions, salad vegetables, pasta, rice and cereals.

food commodities. Therefore the revealed cost of GHG emissions was used as a taxation base in the present analysis, while acknowledging that the costs might vary if valuation of all environmental aspects included were accounted for. Finally, the implications of adjustment of VAT rates, as a way to price foods differently depending on their climate impacts, were investigated (Section 2.1.5).

2.1.1. Taxation based on different sets of food products

As all food production causes climate damage through emissions of GHGs, it can be argued that all foods should be taxed. However, since animal-based food products in general, and beef in particular, cause considerably higher GHG emissions per kg than most plant-based foods (see e.g. Moberg et al., 2019), it can be justifiable to restrict taxation to these products. The administrative costs for running such a system would also decrease.

Another taxation option could be to tax only meat from monogastric animals, i.e. pork and chicken, and eggs, although excluding beef from a tax based on environmental impact is difficult to justify from a climate perspective, considering the high GHG emissions associated with rearing of beef cattle. However, ruminants can contribute positively to food systems, e.g. by maintaining pasture biodiversity by grazing or converting grass and other roughage to food. For these reasons, it can be considered important to sustain ruminant meat production, while limiting consumption of other animal products (e.g. Röös et al., 2016, Van Zanten et al., 2018).

2.1.2. Taxation based on different system boundaries

In a study by Moberg et al. (2019), different ways of calculating the climate impact of foods to be used in food taxation were analysed. Their analysis included the choice of system boundaries, i.e. whether to account for the emissions from the full life cycle of a food product or to choose parts of the life cycle. The full life cycle of a food product includes emissions arising from the production of inputs such as fertilisers, emissions from agricultural production and emissions from post-farm gate activities such as processing, packaging and transportation. Some of these emissions, e.g. from electricity and fuel used in food production, are partly covered by existing tax schemes (e.g. the Swedish tax on CO_2 from energy (Författningssamling, 1994)). Ideally, when applying a climate tax on food, care should be taken to avoid “double-taxation”, in order to achieve a cost-efficient policy instrument (Gren et al., 2019). Moberg et al. (2019) argued that targeting only the emissions arising in the agricultural sector (i.e. the emissions from soils, enteric fermentation and manure management), currently exempted from any tax scheme, would be less complex and ease administration. However, as this might potentially lead to less understanding and acceptance, as well as failing to tax emissions from energy use not covered by current taxation on CO_2 emissions (i.e. for imported products), including other emissions up to farm gate or to retail gate may be important (Moberg et al., 2019).

2.1.3. Taxation based on different weighting of GHG emissions

In policy and product assessments, the total climate impact of different GHGs such as CO_2, methane (CH_4) and nitrous oxide (N_2O), is commonly calculated using Global Warming Potential for 100 years (GWP_{100}) to weight the impact from the different gases into so-called carbon dioxide equivalents (CO_2e). The GWP factors are available both with and without inclusion of the effects of climate-carbon feedback mechanisms, which describe climate impact effects from changes in the carbon cycle (Moberg et al., 2019). Here we simulated the effects of a tax based on both. We also included a weighting of emissions using Global Temperature Potential (GTP) for 100 years (Myhre et al., 2013), as well as GTP factors based on the point in time for which warming of 2 °C according to the Paris agreement is expected to be reached, following the approach of Persson et al. (2015). Persson et al. (2015) argue that such GTP factors are more relevant than a 100-year reference, as they correspond to a time horizon (at some point between 2050 and 2100) when the target is expected to be met.

2.1.4. Taxation based on weighting of several environmental impacts

We also included a scenario where taxation was based on several environmental impact categories, as food production affects the environment in many different ways. Hence, rather than basing the tax solely on the GHG emissions, taxation was also based on other environmental aspects. To weight the different environmental impacts, we used a ‘distance to target’ methodology, where the weighting was based on the extent to which the environmental impact from food consumption exceeded suggested planetary boundaries (Tuomisto et al., 2012). For this, we used the six environmental categories: GHG emissions, cropland use, nitrogen use, phosphorus use, consumptive freshwater use and terrestrial extinction rate. These categories were analysed in the EAT-*Lancet* report by Willett et al. (2019) and environmental boundaries, within which the environmental impacts of the food system should operate to be environmentally sustainable, were established for each aspect. The overall weighted impact (***WI***) for all environmental categories is calculated as (Tuomisto et al., 2012):

$$WI = \sum_i \alpha_i D_i / N_i \tag{1}$$

where α_i is the weighting factor for each impact category (i) (available in Supplementary Table S7), D_i is the impact before weighting in each impact category, and N_i is a normalisation factor, used to normalise the impact in each category to the highest value of a food product within that impact category.

2.1.5. Adjustment of VAT rates

As an alternative to environmental taxation, price changes could be implemented by changes to the existing VAT system. Member states of the European Union are obliged to impose a minimum VAT rate of 15% on goods and services, but reduced rates down to 5% may be applied on goods and services that are seen as particularly necessary for consumers (Swedish Tax Agency, n.d.-b). In Sweden, the current standard VAT rate is 25%, while food products enjoy a rate of 12% and books, cultural events and personal transportation have a VAT rate of 6% (Swedish Tax Agency, n.d.-c). To establish a price difference between meat and dairy products based on their higher climate impacts and plant-based foods, VAT levels can be adjusted for these products. Here, three variants of differentiating VAT levels were investigated; i) applying the standard VAT level of 25% on animal products and keeping 12% for other foods, ii) applying 25% on animal products and a reduction to 6% on fruit, vegetables and cereals, and iii) a reduction to 6% on fruit, vegetables and cereals and 12% on all other foods.

2.2. Estimating consumer response to price changes in food

Säll et al. (2020) calculated price elasticities for 52 food groups using historical price and consumption data on the foods (yearly national average data ranging from 1980 to 2015). An overview of the 52 food groups is provided in Supplementary Table S1, while price and consumption data can be found in Supplementary Table S3 and Table S4, respectively. Consumption data were mainly retrieved from the Swedish Board of Agriculture (SBA, 2019), but also from the FAO (2018). Price data were retrieved from Statistics Sweden (2020).

The demand system described by Säll et al. (2020) was estimated using the Quadratic Almost Ideal Demand System (QAIDS) model, which is a common econometric model for estimating demand elasticities (Deaton and Muellbauer (1980), and used in for example Nordström and Thunström (2009)). The system is estimated assuming ‘weak separability’, which means that the budget for all consumption included is a constant share of the consumer’s total budget before and after price changes (Edgerton, 1997).

Three stages of food groups are included in the demand system (Fig. 1), where the lower level covers each of the 52 food groups, such as beef, pork meat, tomatoes and coffee. These food items are then aggregated into 14 groups at a middle level, according to similarity in

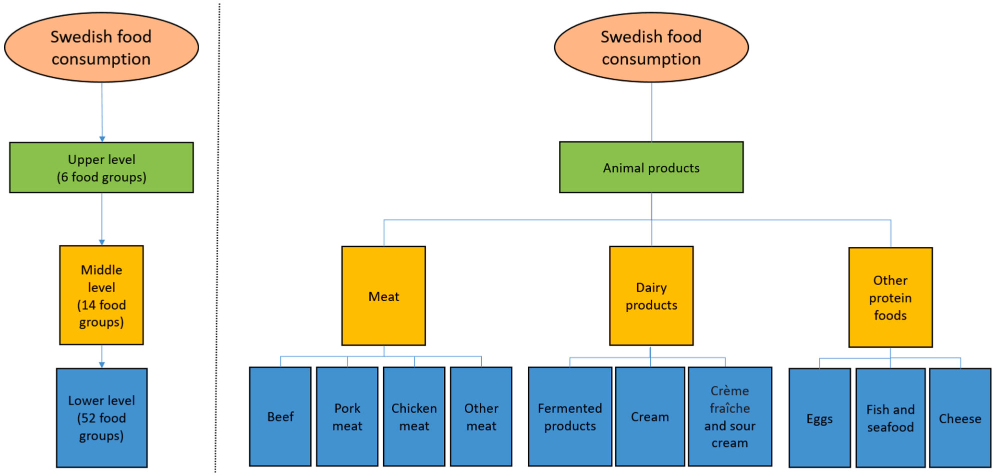

Fig. 1. Example of food commodities and commodity groups included at the three different levels in the demand system.

their function. Fig. 1 exemplifies this for the middle level group of 'Meat', in which all meat products are included. Fermented products and cream are included in the middle level group of 'Dairy products'. In an upper level, both 'Meat' and 'Dairy products' are aggregated into the main group of 'Animal products'. There are in total six main groups at the upper level: animal products; fats; grain and starchy root vegetables; fruit and vegetables; drinks; and snacks.

As described in Säll et al. (2020), elasticities for each of the three levels were calculated and then combined to find final elasticities, showing how consumption of the 52 foods in the lower level changed in relation to each other due to the price changes from taxation (for more details about calculation of elasticities in a multi-stage demand system, see Edgerton (1997)). The largest changes in consumption from a price change will be within the food group on the lower level, i.e. consumers are assumed to first change between products within the same low-level group, e.g. choose pork instead of beef. The model also accounts for whether products are regarded as complements or substitutes. If two foods are viewed as complements, this means that the consumer wants to include both in a varied food basket, while substitutes are regarded as exchangeable to a certain degree. The results of the elasticities and a more thorough method discussion can be found in Säll et al. (2020), while a discussion of how the elasticities compare to other studies is provided in the Supplementary Material to this paper.

2.3. Effects on energy intake of the Swedish population

As a complement to investigating the amount of food consumed as a result of the taxation scenarios, we also investigated how the energy intake of the Swedish population was affected by taxation. For this, we retrieved data from the Swedish National Food Agency (2020a) on the amount of calories in the food products included in the analysis (available in Supplementary Table S5). Using the demand system developed by Säll et al. (2020), we then applied these data to simulate the effects on average calorie intake in the Swedish population arising from applying taxation on all foods on the market.

2.4. Environmental aspects, indicators and inventory data for evaluation of the tax schemes

The focus in this study was on investigating the environmental effects of implementing taxation to reduce the environmental impacts of Swedish food consumption, and hence to evaluate aspects of importance for the Swedish context. Moberg et al. (2020) identified environmental aspects and indicators relevant for assessing the environmental sustainability of Swedish food consumption, building upon the Swedish Environmental Objectives which steer Swedish environmental policy work. Moberg et al. (2020) also considered current obstacles to using the indicators, which include e.g. lack of inventory data for pesticide use and site-dependent impact assessments of e.g. eutrophication. Based on the current limitations discussed in Moberg et al. (2020), we chose the indicators listed in Table 2.

With regard to evaluation of atmospheric aerosols, Moberg et al. (2020) suggested using an indicator including emissions of nitrogen oxides (NO_x) and particles. Here we chose to evaluate this aspect based on NO_x emissions, most of which arise from transportation (both national and international) by road and sea (Statistics Sweden, 2020). Further, as discussed in Moberg et al. (2020), around 90% of global fish stocks are estimated to be fished at levels at or above capacity, and therefore it is of major importance to include aspects of marine extinction in environmental assessments. However, detailed consumption data on fish and seafood are currently lacking in Swedish statistics, so it was not possible to model how price changes would affect consumption of different fish species. Rather, analysis of the effects of taxation only covered the effects on the average consumption of fish and seafood. The indicator marine extinction rate, suggested to evaluate marine biodiversity by Moberg et al. (2020), was therefore excluded from our analysis. Inventory data used for all indicators in this study comprised emissions or impacts caused by production of 1 kg or litre of food. The data aimed at covering impacts caused both by food produced within Sweden and imported food (see Moberg et al., 2019). A summary of the main sources of inventory data is presented in Table 2, while the sources are described in detail in Supplementary Table S1. Environmental data for all food products are provided in the Supplementary Material.

3. Results

Section 3.1 shows the changes in price, quantity and energy intake in the different tax schemes, while Section 3.2 describes the environmental effects. The focus is on results from the taxation scenarios for different sets of food products, weighting of several environmental impacts, and

Table 2
Environmental aspects and indicators chosen for evaluating the effects of taxation, based on Moberg et al. (2020).

Environmental aspect	Suggested indicator for evaluation of the aspect	Description of inventory data and main sources to data
Climate change	GHG emissions	GHG emissions taken from Moberg et al. (2019) and Moberg et al. (2020)
Land-system change	Cropland use	Yield levels for plant-based products and feed taken from Moberg et al. (2019) and Moberg et al. (2020)
	Pasture use	Time on pastures and pasture use per animal taken from Moberg et al. (2019) and Moberg et al. (2020)
Nitrogen and phosphorus cycling	Nitrogen and phosphorus application	Nitrogen and phosphorus from application of mineral fertiliser and nitrogen from biological fixation by plants, taken from Moberg et al. (2019) and Moberg et al. (2020)
Freshwater use	Consumptive freshwater use	Consumptive blue water use (groundwater and surface water) for crop irrigation and rearing of animals, which reduces the flows in watersheds as it does not flow back to the same river or aquifer. Data taken from Moberg et al. (2020).
Biodiversity loss	Terrestrial extinction rate	Based on the potential endemic species loss of five taxa (mammals, birds, reptiles, amphibians, plants) from occupation of cropland and pastures in different countries. Data taken from Moberg et al. (2020).
Atmospheric aerosols	NO_x emissions	Emissions calculated through the NTMCalc Environmental Performance Calculator (NTM, n.d.), see Supplementary Table S1 for details.
Acidification of freshwater and land	Ammonia (NH_3) emissions	Emissions from application of mineral and organic fertiliser to fields, direct storage of manure, and losses from ventilation in barns. Data on fertiliser application rates and direct deposition of urine and manure on pasture taken mainly from Moberg et al. (2019) and Moberg et al. (2020). Emission factors for the resulting NH_3 emissions were obtained from the Swedish National Inventory Reporting. Emissions data for storage of manure and losses from ventilation in barns were taken from Moberg et al. (2019).
Chemical pollution	Pesticide use	Based on the amount of active ingredient in the pesticides taken from different sources in the following order of prioritisation, based on availability: country-specific statistics; country-specific data through guidelines or advisory services; country-specific data from the European Union of the average use of different crops or crop categories in the member countries; country-specific or average data from the Ecoinvent database (Ecoinvent Centre, 2019).
Ozone depletion	N_2O emissions	Emissions of N_2O taken from Moberg et al. (2019) and Moberg et al. (2020)

adjustment of VAT rates. Detailed results of all tax scenarios described in Table 1 are presented in the Supplementary Material.

3.1. Changes in price, quantity and energy intake in the different tax schemes

The percentage price changes in different taxation scenarios for an illustrative set of foods are shown in Fig. 2A, while Fig. 2B shows the contribution of different foods to the resulting quantity changes. Table 3 summarises the changes in quantity and energy intake.

3.1.1. Taxation based on different set of food products

In the scenarios taxing different numbers of food products, price changes were the same for the products included, as they were all based on the GHG emissions caused during production, for which the same marginal cost per kg emissions was applied. The relative price changes for different foods depended on both the climate impact of the individual foods and their initial price. The highest percentage change in price was seen for meat and dairy products, especially beef and cheese (Fig. 2A), which is mainly explained by their high climate impact. Price changes for plant-based products were lower, with the exception of products such as coffee and rice, which have a higher climate impact per kg than other plant-based products. For cereals, the percentage price change was relatively high, which is mainly explained by their low initial price.

With regard to changes in quantities consumed, a net decrease in overall food consumption ranging from 22 to 144 g/day (-1.1% to −7.4% of current daily consumption levels of 1942 g) was seen for all scenarios taxing different numbers of food products. This corresponded to a change in energy intake of between 16 and 172 kcal/day (Table 3). Including all products or all animal products in taxation resulted in a large decrease in intake of many animal products, such as milk, cheese, beef and chicken. Furthermore, a large decrease was seen for non-alcoholic beverages, e.g. fizzy drinks and cider. In the demand system by Säll et al. (2020), cold drinks were found to be strong complements to meat, so a price increase for meat also resulted in a decline in cold drink consumption. This finding should be viewed with caution and is a result of how demand parameters are restricted in the system.

In the scenarios targeting only beef or where the taxation covered only monogastric meat and eggs, similar quantity changes were seen as for the scenarios targeting all products or only animal products, i.e. with a large decrease in many animal products. However, focusing taxation on beef resulted in a greater decrease in beef consumption (25% of overall consumption changes, compared with a 7.4% reduction in the scenario targeting all products). Focusing taxation on monogastric meat and eggs resulted in a large decrease in consumption of chicken (40% of overall changes, compared with 6.9% of the changes in the scenario targeting all products).

The more products included in a tax scheme, the higher the cost of food and thus the larger the overall decrease in food consumption. However, although there was a net decrease in overall food consumption, the quantities of some food products or groups still increased. Implementing taxation on all food products increased consumption of sugar and sweeteners (Fig. 2B). Taxation targeting only animal products resulted in a small increase in consumption of products such as potatoes and bread. Restricting the tax to beef increased consumption of primarily bread, cheese, pork and chicken, while targeting monogastric meat and eggs increased consumption of primarily cheese and other meat (i.e. sheep meat, game meat and offal).

3.1.2. Taxation based on weighting of several environmental impacts

With taxation based on a weighted score of several environmental impacts, higher percentage price changes were found for products such as vegetable oil, coffee and cocoa compared with when taxation was based on the climate impact only (Fig. 2A). This is explained by the relatively high impact of these products in environmental categories other than GHG emissions, such as extinction rate and freshwater use (Moberg et al., 2020). Despite the price differences in the weighting scenario, the distribution of quantity changes for different foods was similar to that seen with taxation based only on the climate impact of all foods. The greatest decrease was thus seen in consumption of beef, chicken, cheese and milk, as well as non-alcoholic beverages (Fig. 2B). In this scenario, a net decrease in food consumption was seen, of 71 g in total or a 3.6% reduction in current daily consumption levels, corresponding to a decrease in energy intake of 86 kcal per day (Table 3). Further, as was the case for the scenario targeting all products, a slight increase was found for consumption of sugar and sweeteners (Fig. 2B).

3.1.3. Adjustment of VAT rates

Increased VAT rates for animal products (from 12% to 25%) resulted in a net decrease in food quantities consumed of 109 g/day (-5.6% of current daily consumption), corresponding to a decrease of 111 kcal/

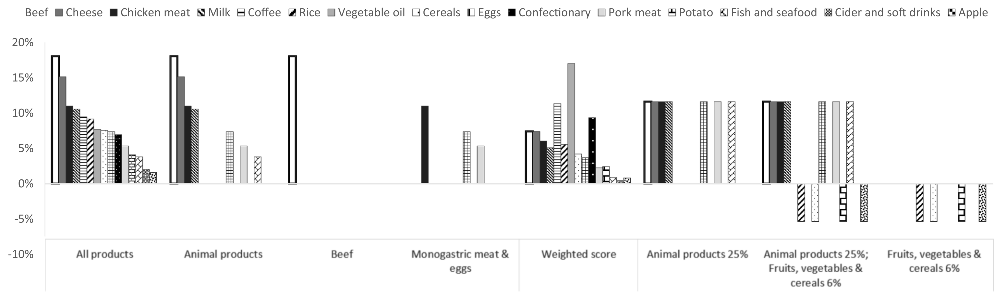

Fig. 2A. (above). Percentage price changes per kg or liter product due to taxation in different scenarios for a set of food products.

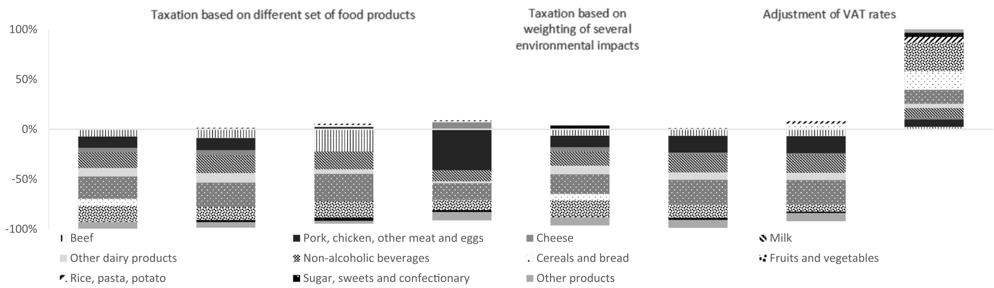

Fig. 2B. (below). Distribution of quantity changes due to taxation in different scenarios for all foods.

Table 3
Changes in quantity and energy intake per capita and day due to taxation in different scenarios.

	Taxation based on different sets of food products				Taxation based on weighting of several environmental impacts	Adjustment of VAT rates		
	All products	Animal products	Beef	Monogastric meat and eggs	Weighted score	Animal products 25%	Animal products 25%; Fruit, vegetables and cereals 6%	Fruit, vegetables and cereals 6%
Changes in quantity (g/capita and day)	−144	−103	−31	−22	−71	−109	−82	27
Changes in energy intake (kcal/ capita and day)	−172	−116	−28	−16	−86	−111	−75	36

day (Table 3). The greatest reductions in quantities consumed were found for the products assigned a higher VAT rate, i.e. animal products, together with non-alcoholic beverages. Similar quantity changes were observed when simultaneously increasing VAT rates for animal products and decreasing the rates for fruit, vegetables and cereals. In both of these scenarios, there was a slight increase in consumption of products such as potatoes, cereals and bread (Fig. 2B). Reduced VAT rates for fruit, vegetables and cereals (from 12% to 6%) resulted in an increase in consumption of 27 g (+1.4% of current daily consumption), or 36 kcal (Table 3). Here, the largest part of the consumption increase was for fruit, vegetables and cereals, and for milk and non-alcoholic beverages.

3.2. *Environmental effects of taxation*

The environmental effects resulting from the different tax schemes are shown in absolute values in Table 4 and as percentages in Fig. 3.

3.2.1. *Taxation based on different set of food products*

For the three scenarios of taxing all products or only including animal products or beef in the taxation, there was a reduction in the environmental impacts for all categories (Table 4 and Fig. 3). The greatest effects were seen for the scenario including all products, since when more products are taxed, less money is available and overall consumption decreases. In this scenario, a reduction in GHG emissions of 1.9 Mton was seen (Table 4), corresponding to about 10% of current emissions (Fig. 3). Large reductions were also seen for pasture use (0.31 Mha, or 12% of current use). In the scenario where only monogastric meat and eggs were taxed, the absolute effects on climate impact and on many other environmental categories were small (e.g. −0.14 Mton of GHG emissions, corresponding to 0.73% of current emissions).

With regard to the relative effects between climate impact and other environmental categories in the scenario targeting only monogastric meat and eggs, these differed more radically from the other scenarios, as both pasture use and extinction rate can be expected to increase. This is

Table 4
Environmental effects from taxation in the different tax schemes.

	Environmental effects from taxation based on different sets of food products	Environmental effects from taxation based on weighting of several environmental impacts	Environmental effects from adjustment of VAT rates					
	All products	Animal products	Beef	Monogastric meat and eggs	Weighted score	Animal products 25%	Animal products 25%; Fruit, vegetables and cereals 6%	Fruit, vegetables and cereals 6%
GHG emissions (Mton)	−1.9	−1.6	−0.73	−0.14	−0.98	−1.4	−1.2	0.20
Cropland use (Mha)	−0.28	−0.22	−0.087	−0.029	−0.16	−0.19	−0.16	0.031
Pasture use (Mha)	−0.31	−0.27	−0.17	0.024	−0.16	−0.21	−0.18	0.024
N application (kton)	−57	−47	-22	−2.6	−30	−38	-33	5.2
P application (kton)	−3.4	−2.5	−0.90	−0.57	−2.2	−2.4	−2.0	0.47
Consumptive freshwater use (Mm3)	−26	−19	−7.1	−2.6	−15	−18	−14	4.5
Terrestrial extinction rate (E/MSY)	−0.0045	−0.0029	−0.0015	0.00045	−0.0036	−0.0033	−0.0025	0.00083
NO_x emissions (kton)	−0.43	−0.27	−0.11	−0.049	−0.25	−0.29	−0.17	0.11
NH_3 emissions (kton)	−7.3	−6.1	−3.3	−0.30	−3.7	−5.0	−4.4	0.61
Pesticide use (kton active ingredient)	−0.45	−0.31	−0.12	−0.066	−0.35	−0.31	−0.24	0.069
N_2O emissions (kton)	−1.4	−1.1	−0.49	−0.12	−0.71	−0.96	−0.83	0.14

mainly explained by the rise in consumption of other meat products and especially sheep meat (Fig. 2B), where current consumption has a high negative biodiversity impact (Moberg et al., 2020). However, these results are highly sensitive to the production region, as further discussed in Section 4.2. The relative effects for climate impact and the other environmental categories were similar in the scenarios of taxation on all products or only animal products or beef. One exception was larger effects on pasture use with a tax solely targeting beef, explained by the decrease in quantity of beef consumed (Fig. 2B). A decline in pasture area can be considered both positive and negative depending on the production region, as further discussed in Section 4.3.

3.2.2. *Taxation based on weighting of several environmental impacts*

Basing taxation on weighting of several environmental impacts resulted in a reduction in the impacts for all environmental categories (Fig. 3), e.g. a 0.98 Mton reduction in GHG emissions (-5.2% of current emissions from food). The impacts in many environmental categories were similar to the effects of GHG emissions, e.g. 0.16 Mha reductions of cropland use, corresponding to −5.4% of current cropland use. In general, the effects resembled those in the scenario with taxation based only on the climate impact.

3.2.3. *Adjustment of VAT rates*

For the scenarios of increasing VAT rates for animal products (from 12% to 25%) and either keeping other foods at the current level (12%) or simultaneously decreasing VAT for fruit, vegetables and cereals (from 12% to 6%), there was a reduction in the environmental impacts for all environmental categories (Fig. 3). This is explained by the decrease in consumption of high-impact animal products such as beef (Fig. 2B). Increasing VAT rates for animal products reduced GHG emissions by 1.4 Mton (-7.2% of current emissions), whereas a reduction of 1.2 Mton (-6.2%) was found for the scenario with a simultaneous decrease in VAT for fruit, vegetables and cereals. Large effects in both scenarios were also seen for pasture use (0.21 Mha, corresponding to −8.1%, compared with 0.18 Mha, equivalent to −7.2%) and NH_3 emissions (-5.0 kton, corresponding to −8.0%, compared with −4.4 kton, equivalent to −7.0%). The opposite was seen for the scenario with lowered VAT only, where environmental impacts can be expected to increase due to the overall increase in food consumption (Fig. 3). In this scenario, the GHG emissions increased by 0.20 Mton (1.1%). Although similar quantity changes were observed in the scenarios with increased VAT rates for animal products, the scenario with a simultaneous decrease for fruit, vegetables and cereals resulted in a slightly higher increase in consumption of some products, such as potatoes, cereals and bread (Fig. 2B). However, as these products in general have a low environmental impact per kg, the environmental effects were comparable between the scenarios (Fig. 3). For the lowered VAT scenario, the effects in many environmental categories were similar to those for GHG emissions, with the exception of emissions of NO_x which had around 1.9-fold the effects of GHG emissions (Fig. 3). The latter is explained by higher consumption of certain plant-based products that are imported to a large extent or exclusively, and hence require larger transportation distances, leading to increased NO_x emissions.

4. Discussion and policy implications

4.1. *General discussion and policy implications of the results*

Most of the analysed taxation scenarios resulted in a reduction in both the climate impact and other environmental aspects, to a large extent explained by the overall reduction in consumption of foods, and in particular of many animal products. These results are in line with earlier findings, by e.g. Poore and Nemecek (2018), Aleksandrowicz et al. (2016) and Martin and Brandão (2017), that reducing consumption of animal-based food offers potential for simultaneous decreases in climate impact and other environmental burdens.

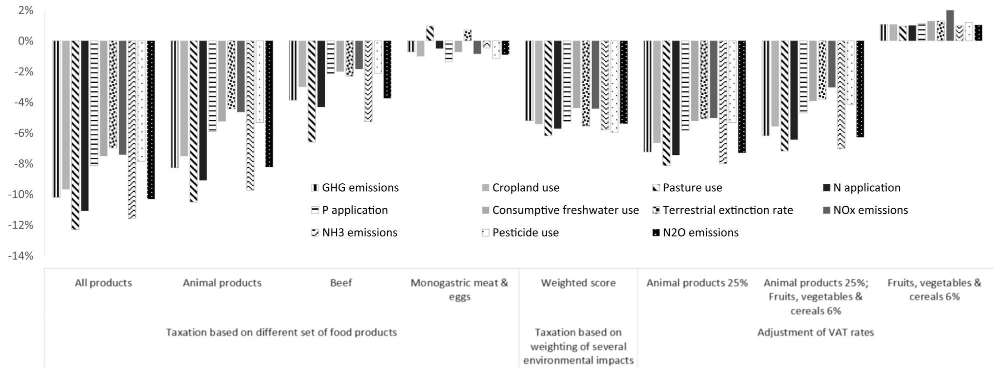

Fig. 3. Change in percentage of environmental effects in the different tax schemes, relative to the current impact of Swedish food consumption.

The largest overall reduction in climate impact and other environmental categories was seen for the scenario in which taxation was implemented on all products, followed by the scenario in which taxation was targeted only at animal products (Fig. 3).

The reductions in climate impact were similar to those reported in previous modelling studies of climate taxation by e.g. Wirsenius et al. (2011), Säll and Gren (2015) and Springmann et al. (2016). Thus environmental taxation offers potential for decreasing climate impact and other environmental aspects. However, on comparing the resulting environmental impacts of food consumption for six of the environmental categories in the scenario of taxation on all food products against the environmental boundaries suggested by the EAT-*Lancet* report by Willett et al. (2019) (Table 5), it is clear that the impacts would be high even after taxation. The only environmental aspect that would not exceed the environmental boundaries would be consumptive freshwater use, with both current consumption and levels after taxation being below the suggested boundary (Table 5). The other environmental impacts would range between 1.7-fold the boundary for cropland use to more than 5-fold the boundary for extinction rate. The environmental boundaries suggested by Willett et al. (2019) do not capture all environmental categories assessed in this study but, as pointed out by Moberg et al. (2020), no environmental threshold for food-related impacts has yet been set for these. Environmental boundaries for indicators such as GHG emissions and cropland use (which are also included in the EAT-*Lancet* report) have been proposed in other studies (see e.g. Röös et al. (2015)), with similar estimates.

The results thus indicate that environmental taxation on food cannot be used as a stand-alone policy for transforming the food system or for curbing the impacts from food consumption. As previously mentioned, both production- and consumption-side measures are necessary to achieve profound changes in the food system for reduced environmental impacts. Within the actions to steer towards consumption-side changes and to achieve dietary changes, food taxation could be one of several public policies implemented in a policy package, where other consumption-side measures could include e.g. information campaigns and 'negative' labelling of high-burdening foods (see e.g. Röös et al. (2020)). Such a policy package should preferably also include policy directed towards the supply chain, i.e. food industry and retail. The retail sector could be required to report and improve on a set of key performance indicators related to the environmental impacts of food sold (see e.g. indicators suggested by the Plating Up Progress project (Food Climate Research Network and Food Foundation, 2019)). Such regulatory systems for reducing emissions are already used in other sectors, e.g. in sales of new cars in the European Union, for which regulations are set on a maximum of GHG emissions per km (EU, 2019).

The scenario involving increased VAT rates on animal-based foods resulted in similar reductions in the environmental burdens as the scenario involving taxation of only animal-based products. Making changes to the existing VAT system would probably be less complex in terms of administrative efforts than implementing a new tax system, as would be the case with a climate tax. Hence, using the existing VAT system might reduce administrative barriers to implementing taxation to steer food consumption in the desired direction of reducing its environmental impacts. However, internalising the climate impact of food production through increasing the VAT rate on high-impacting foods (i.e. animal products) is not a cost-efficient policy measure in a strict sense, as it requires emissions from all sources (food, energy, transport etc.) to be taxed alike (Baumol and Oates, 1988). Hence, the option of implementing climate taxation in Sweden, based on the climate impact of food with the same cost used in the Swedish tax on CO_2, would be a more cost-efficient option.

With regard to the scenario involving decreased VAT rate on fruit, vegetables and cereals, the results indicated an increase in overall food consumption (Fig. 2B), and thus imposed greater burdens in all environmental categories (Fig. 3). Similar results were found by Broeks et al. (2020), although they pointed out that increased consumption of plant-based foods could still lead to a net societal benefit due to e.g. reduced healthcare costs. Simultaneously increasing the VAT rate on animal-based foods and lowering it for fruit and vegetables resulted in overall reductions in both the climate impact and other environmental aspects (Fig. 3). While the results are less pronounced than in the scenario where the VAT rate was only increased on animal-based foods, it is plausible that such a simultaneous change could lead to higher acceptance than targeting only animal-based products, as consumers would be compensated (Röös et al., 2020).

4.2. Effects of taxation on extinction rate

In the scenario targeting monogastric meat and eggs, an increase was seen in consumption of sheep meat, which resulted in a rise in extinction rate (Fig. 3). As discussed in Moberg et al. (2020), the biodiversity impact is highly dependent on the production region. Sheep meat produced in New Zealand currently accounts for 20% of Swedish market share and causes much higher biodiversity loss from land occupation than sheep meat produced in Sweden. However, if all sheep meat were to be produced in Sweden when consumption increased, the scenario targeting monogastric meat and eggs would result in a decrease in biodiversity impact (Supplementary Table S11). Although this taxation scenario would be difficult to justify due to the exclusion of the highly climate-impacting ruminant products, the results are interesting as they

Table 5
Environmental impacts per capita resulting from taxation compared with the suggested environmental boundaries from the EAT-Lancet report (Willett et al., 2019).

	GHG emissions (ton CO2e)	Cropland use (ha)	N application (kg)	P application (kg)	Consumptive freshwater use (m3)	Terrestrial extinction rate (E/MSY)
Current environmental impacts per capita (results from this study)	2.0	0.30	54	4.4	37	6.7×10^{-9}
Environmental impacts per capita with taxation on all food products (results from this study)	1.8	0.27	48	4.0	34	6.3×10^{-9}
Per capita boundary (downscaled from the global boundaries given by the EAT-*Lancet* Commission)	0.68	0.18	12	1.08	50	1.4×10^{-9}

illustrate the sensitivity in the results to the region where production will increase. This implies that there could be a potential goal conflict between reducing the climate impact and the extinction rate if taxation results in continuation or expansion of production of foods in more vulnerable regions.

4.3. Effects of taxation on pasture use

As can be seen in Fig. 2B, many of the taxation scenarios resulted in decreased consumption of beef as well as sheep meat, which led in turn to a decline in pasture use (Fig. 3). This could be considered positive, as land clearance for pasture is currently a major driver of deforestation and other land use change. Further, environmental targets on land use, such as the EAT-*Lancet* boundary on land system change, are based on limiting further expansion of agricultural land globally (Willett et al., 2019). However, in Sweden and other countries in Europe and globally, intensification and restructuring of agriculture have led to fewer grazing animals, which poses a threat to biodiversity-rich semi-natural pastures. In addition, many threatened species in Sweden are linked to traditional agricultural landscapes. As a consequence, the national environmental objectives for land use aim at maintaining, rather than decreasing, Swedish agricultural land and especially at preserving semi-natural pastures (Moberg et al., 2020).

In the scenarios where a reduction in pasture use was seen, a maximum decline of about 310,000 ha pasture land was found for the scenario with taxation on all food products, corresponding to 12% of current pasture use. Note that this is based on average pasture area per animal. Due to limitations in calculation of consumer demand in the present study, it was not possible to determine whether this decline derived from meat produced in Sweden (Swedish beef and sheep meat currently represent 53% and 33% of the market, respectively), or whether the animals grazed semi-natural pastures or cropland, managed grasslands and/or semi-natural pastures.

Assuming an extreme scenario (i.e. with taxation on all food products), where the entire reduction derived from meat produced in Sweden, with animals grazing solely on semi-natural pastures, this would entail a reduction of up to 70% of the current area of 450,000 ha semi-natural pastures in Sweden. Depending on consumer response to the tax, this could introduce a goal conflict with maintaining these sensitive areas. However, considering the current trend for consumers to favour Swedish meat over imported meat (SBA, 2020), such an extreme outcome is unlikely.

Reduced consumption levels after introduction of a tax would result in yearly per capita consumption of around 22 kg beef (carcass weight), much of which is imported and involves grazing on managed grasslands or cropland or zero-grazing, where the animals are kept indoors for most of the year (Moberg et al., 2020). In comparison, a study by Röös et al. (2016) found that maintenance of the current area of semi-natural pastures in Sweden could be compatible with reducing per capita consumption of beef meat to 4–14 kg (3–10 kg of bone-free meat, depending on different intensity levels) per year. Further, Larsson et al. (2020) found that there is no shortage of ruminant animals to maintain Swedish semi-natural pastures, but rather that the animals are housed for long periods of the year, including during the grazing season, or that they graze on grass-clover leys on cropland. This is due to the high cost to farmers of rearing their animals on semi-natural pastures in comparison with other production systems. A conclusion from Larsson et al. (2020) was therefore that more targeted policy instruments are needed for maintenance of semi-natural pastures, e.g. increased payments to farmers for management of these areas using grazing animals.

In summary, consumption of beef could be reduced to a much greater extent than achieved through the taxation scenarios simulated in the present study, without creating a lack of ruminant animals for grazing semi-natural pastures. To avoid reductions in biodiversity in semi-natural pastures, policy instruments should be introduced on the production side to stimulate this type of production, e.g. by giving larger payments to farmers who keep their animals on semi-natural pastures. This could be achieved by using the income from taxation on the consumption side, which could also lead to increased acceptance of taxation (Kallbekken and Sælen, 2013, Bachus et al., 2019).

4.4. Effects of taxation on use of pesticides

Most taxation scenarios resulted in a net decrease in food consumption, but with a slight increase in consumption of some products such as potatoes, cereals and bread in the scenarios with taxation targeting animal products, or when simultaneously increasing VAT rates for animal products and decreasing the rates for fruit, vegetables and cereals. Further, for the scenario with decreased VAT rates on fruit, vegetables and cereals, a net increase in consumption was seen, in particular for the products with decreased VAT rate. Earlier studies have shown increased toxicity effects from increased consumption of certain plant-based foods (e.g. Martin and Brandão, 2017). In the present study, chemical pollution was analysed through the indicator pesticide use (amount of active ingredient), but the toxicity of different pesticides can vary by several orders of magnitude. However, lack of data (e.g. on the types and amounts of pesticides used in different crops) made it impossible for us to assess the actual impacts on ecosystems (Cederberg et al., 2019). Hence, it is possible that increased consumption of certain plant-based foods could lead to increased toxicity. This would be the case for the scenario with decreased VAT rates, where overall consumption of food was seen to increase, with an overall increase in the use of pesticides. With regard to the scenarios where an increase in consumption of potatoes, cereals and bread was seen, consumption of beef, pork and chicken was found to decrease. This would lead to a net reduction in consumption of cereals, due to the decrease in use of cereals as feed. Hence, the marginal effects on ecotoxicity would probably not increase in these scenarios.

While plant-based foods may cause toxic effects on ecosystems, an assessment by Nordborg et al. (2017) of freshwater toxicity due to pesticide use in food production found that animal-based foods, especially pork and chicken, cause considerably higher ecotoxicity in freshwater than legumes and cereals. In the present study, an increase was seen in consumption of pork and chicken in the taxation scenario targeting only beef. While this did not result in more extensive use of pesticides calculated as amount of active ingredient, we tested whether this might cause higher freshwater ecotoxicity. We applied impact factors for beef, milk, pork and chicken from Nordborg et al. (2017) to the changes in quantity resulting from the taxation scenario targeting beef. We also tested the toxicity impacts per kg feed product from Nordborg et al. (2017) for the feed rations used in this study. The results are shown in Supplementary Table S12. For both, taxation resulted in a marginal decrease in freshwater ecotoxicity impacts. However, the results indicate a higher freshwater ecotoxicity when applying the factors per kg food (of beef, milk, pork and chicken), than when using the toxicity factors per kg feed with the feed rations used in this study. This is explained by the higher amount of soy in the feed rations used in pork and chicken production in Nordborg et al. (2017), for which the freshwater ecotoxicity was notably higher than for other feed products. When those authors calculated the freshwater toxicity for pork and chicken with soy-free feed rations, the values were found to be reduced by 70 and 91%, respectively.

In summary, no increase was seen in the toxicity effects due to the increase in consumption of pork and chicken meat. However, toxicity effects might increase with a larger rise in consumption of pork and chicken, if these originate from production systems using substantial amounts of soy.

4.5. Potential social and economic goal conflicts

Apart from goal conflicts between different environmental aspects, potential social and economic goal conflicts could also arise from taxation, e.g. concerning nutrient intake of the population and distributional effects for consumers. With regard to potential goal conflicts to nutrient intake in the population, taxation in most scenarios showed an overall decrease in food consumption, resulting in reductions in energy intake of up to 172 kcal per capita and day (Table 3). According to the latest dietary survey, current calorie intake from Swedish food consumption is approximately 2800 kcal per capita and day at population level (with underestimations of ~ 6–9%), while the average recommended intake is between 1700 and 3200 kcal per day, depending on age, sex and level of physical activity (Swedish National Food Agency, 2020b). In the present analysis, calorie intake after taxation was approximately 2600 kcal in the scenario with the largest reductions in food consumption. In the scenario with reduced VAT rates on fruit, vegetables and cereals, an increase was seen in food consumption, which resulted in an increase in energy intake of 36 kcal per capita and day. While we acknowledge that ours was a simplified analysis made at population level and only including caloric intake, the results indicated that a climate tax would only lead to small differences in energy intake at population level, without leading to either insufficient or exceeded recommended energy intake.

Concerning distributional effects, Säll (Manuscript in review) found that a climate tax implemented on all food products on the Swedish market would be regressive, with the largest impact on e.g. families with children and the unemployed. To balance such goal conflicts, those authors point out the necessity of implementing policies to compensate consumers when introducing food taxation.

4.6. Limitations and uncertainties

4.6.1. Estimations of consumer response to price changes in food

Calculations in the demand system developed by Säll et al. (2020) are restricted to the level of detail of the commodity groups provided in available national statistics. For example, no distinction is made in the Swedish statistics between production countries or production systems, e.g. between organic and conventional food production. Hence, it is not possible to distinguish the consumer response to such specific commodity groups. Furthermore, during the past few years there has been an increase in consumption of plant-based meat alternatives (Zachrisson, 2019), but consumption and price data are lacking for these products, so they could not be included in the demand system in this study. If taxation leads to decreases in consumption of meat and dairy products but simultaneous increases in consumption of plant-based alternatives with e.g. nuts, coconut and palm oil, there could be a risk of increased environmental burdens of e.g. biodiversity loss, freshwater consumption and pesticide use, depending on the production region. The risk of such substitution (not seen from the demand system used here) is an important topic for further studies. More information about the demand system and its limitations are available in Säll et al. (2020).

4.6.2. Estimations of the environmental impact of foods

As previously mentioned, the results of environmental impact assessments can differ depending on the choice of indicator and whether the impacts are global or site-specific. For example, nitrogen and phosphorus application and freshwater use may serve as proxies for the risk of eutrophication and water scarcity, respectively, but do not capture local aspects such as nutrient status of recipients or water availability (Moberg et al., 2020). Further analysis on this issue is currently hampered by lack of detailed inventory data and by difficulty in predicting where production would be altered and how changes in production systems would affect local waterways. Hence, it is possible that taxation could lead to site-specific impacts which are not captured by the current indicators.

The marine extinction rate of fish and seafood could not be analysed, due to lack of data. In the scenario targeting only beef, a 0.33% increase in consumption of fish and seafood was seen, whereas a 0.95% increase was found for the scenario with decreased VAT rates on fruit, vegetables and cereals (Supplementary Table S4). There is a potential risk that the pressure on marine extinction rate would be exacerbated by this increase, if the increase in consumption involves vulnerable species. However, consumption of some species might decrease, so the net burden could remain the same.

Much of the data on the environmental impacts of foods used in this study was taken from earlier studies, such as those by Moberg et al. (2019) and Moberg et al. (2020), where limitations and uncertainties are discussed. For example, as discussed in Moberg et al. (2020), calculation of environmental impacts is straight-forward for some environmental indicators, e.g. the cropland use indicator is calculated using data on crop productivity levels, for which data availability is generally good. For other indicators such as GHG emissions, calculations depend on crop productivity levels plus a range of other variables, many of which fluctuate due to factors such as climate conditions and soil characteristics. Further, detailed inventory data may be lacking for some variables, and especially for countries outside Europe. To account for variations, the calculations in Moberg et al. (2019) were primarily based on an average of five years for influential variables such as slaughter statistics, yield levels, fertiliser application rates and energy use for heating greenhouses. Further, the calculations were based on a weighted average of different production countries reflecting the market shares in Sweden.

The data used in calculations of environmental impacts in this study, e.g. on pesticide use and ammonia emissions, also include uncertainties. For example, statistics on pesticide use are provided for some countries, although often for crop groups rather than specific crops. Further, recent data on pesticide use are lacking in several countries, including the European Union, where data were last published in 2007 (Eurostat, 2007). With respect to calculations of ammonia emissions, this required inventories of application rates of mineral and organic fertiliser to fields, as well as emission factors for the fertiliser products used. As discussed in Moberg et al. (2020), data on application rates of fertilisers to some crops are available from statistical databases and advisory services in Sweden and other European countries, while such site-specific data are lacking for certain crops or production countries, especially countries outside Europe.

Building on work by Moberg et al. (2019, 2020), the environmental impact data used in this study were based on the average emissions and resource use directly associated with current production of foods. These data were also used in estimation of the environmental impact from taxation. Hence, the environmental impacts due to changes in consumption patterns identified with the demand system by Säll et al. (2020) were accounted for. However, estimation of the effects from taxation were based on the current average environmental impact, rather than the marginal impact. A marginal approach would require more detailed data of where, and with what production technologies, production of the foods would increase or decrease as a consequence of taxation, which the current demand system does not provide. Based on the discussion above, there are a number of uncertainties associated with the environmental impact results in the present study, but the task of estimating these uncertainties is currently hampered by lack of available data on the uncertainties themselves, or by large variations in the variables. While it would be possible to provide gross uncertainty ranges for e.g. cropland use by using data on variability in crop yield levels, it would be an immense task to establish uncertainties for other indicators used in this study, such as GHG emissions, and for all food products included in the study.

For a wider discussion on the uncertainties and limitations of the data on nitrogen and phosphorus application, freshwater use and estimation of biodiversity loss, see Moberg et al. (2020).

4.7. Summary of effects of taxation and policy implications

The effects of the different tax scenarios on environmental impacts are summarised in Table 6, which indicates whether taxation would result in an increase or decrease in the absolute value of each environmental indicator and the risk of potential goal conflicts related to the environmental indicators. Identification of potential goal conflicts from taxation is important when designing policies and policy packages, in order to avoid or balance the potential negative aspects resulting from taxation, as discussed in Section 4.1.

With regard to nitrogen and phosphorus application, freshwater use and emissions of NO_x, NH_3 and N_2O, no evident goal conflicts from

Table 6
Summary of effects of the different tax schemes on environmental impacts, where green panels indicate a decrease in the absolute effect of the environmental indicator, red panels indicate an increase, and orange panels indicate a risk of goal conflicts, which should be taken into consideration in policy development.

Environmental aspect (indicator)	**Taxation based on different sets of food products**				**Taxation based on weighting of several environmental impacts**	**Adjustment of VAT rates**		
	All products	Animal products	Beef	Monogastric meat and eggs	Weighted score	Animal products 25%	Animal products 25%; Fruit, vegetables and cereals 6%	Fruit, vegetables and cereals 6%
Climate change (GHG emissions)								
Land-system change (Cropland use)								
Land-system change (Pasture use)								
Nitrogen and phosphorus cycling (N and P application)								
Freshwater use (Consumptive freshwater use)								
Biodiversity loss (Terrestrial extinction rate)								
Atmospheric aerosols (NO_x emissions)								
Acidification of freshwater and land (NH_3 emissions)								
Chemical pollution (Pesticide use)								
Ozone depletion (N_2O emissions)								

taxation related to these environmental categories were identified in the present analysis. However, as discussed in Section 4.6.2, taxation could potentially lead to site-specific impacts which are not captured by the current indicators.

5. Conclusions

Most taxation scenarios analysed gave a net decrease in food consumption, reducing the burden for climate change and other environmental impact categories. An exception was a scenario assuming reduced VAT rates for a selection of plant-based foods, where a net increase in food consumption was seen, resulting in a higher burden for all environmental categories. Many of the scenarios resulted in decreases in meat and dairy products, owing to a high percentage price change due to their high climate impact. A reduction in beef consumption would cause a decline in pasture use, which is positive from a global perspective by limiting further expansion of agricultural land. From a Swedish perspective, reducing consumption of beef could potentially create a goal conflict with maintaining biodiversity in semi-natural pastures. To avoid this, meat produced on semi-natural pastures could be further incentivised by production-side measures. With regard to biodiversity loss (modelled as extinction rate of terrestrial species), the overall burden could increase as a result of taxation if consumption of products from biodiversity-rich regions were to increase.

CRediT authorship contribution statement

Emma Moberg: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Resources, Software, Validation, Visualization, Writing - original draft, Writing - review & editing. **Sarah Säll:** Conceptualization, Methodology, Validation, Writing - review & editing. **Per-Anders Hansson:** Conceptualization, Methodology, Supervision, Validation, Writing - review & editing. **Elin Röös:** Conceptualization, Funding acquisition, Methodology, Project administration, Supervision, Validation, Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

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Climate taxation on foods has been suggested to change eating patterns and reduce environmental impacts from food consumption. This thesis offers insights into how data on environmental pressures of foods can be calculated for application in taxation and assessments of the sustainability of diets. The work included assessing the environmental sustainability of Swedish food consumption and identifying missing aspects for capturing the environmental sustainability of the diet in a Swedish context. Potential goal conflicts resulting from taxation were also identified.

Emma Moberg received her postgraduate education at the Department of Energy and Technology, Swedish University of Agricultural Sciences (SLU), Uppsala. She holds a Master of Science degree in Environmental and Water Engineering from Uppsala University.

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