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The true cost of Swedish food consumption

A true cost accounting assessment of externalities associated with food consumption in Sweden

Bachelor's thesis in Global Systems Engineering

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Cover: AI-generated illustration depicting a dinner plate with Earth placed at its center

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Abstract

This thesis focuses on estimating the true cost of food consumption in Sweden by assessing the environmental and health related externalities associated with various food categories. The market fails to account for these costs, leading to unsustainable consumption patterns and significant societal burdens. This study employs a True Cost Accounting approach to quantify and monetise various externalities, including global warming, eutrophication, degradation of ecosystems, biodiversity loss, ecotoxicity, human toxicity, and disease burdens. By utilising data from the Sustainability Assessment of Foods and Diets database and the Global Burden of Disease study, the research provides a comprehensive overview of the environmental and health impacts of food consumption in Sweden.

The findings reveal a total external societal cost of SEK 301 billion, with 81% stemming from health-related externalities and the remaining 19% from environmental. When these costs are taken into account, the true cost of Swedish food consumption is nearly double the current national food expenditure. Externalities following dietary risks and carbon emissions from food production are the largest contributors. Meat-based food categories contribute significantly to both environmental degradation and health-related costs, while other categories such as wholegrain pasta and vegetables would generate a societal net benefit per kilogram due to their positive health impacts. The thesis also discusses the potential of market based interventions, such as taxes and subsidies, to internalise these costs and promote a more sustainable and healthy food consumption in Sweden. By incorporating these externalities into food pricing, this study aims to provide insights that guide policymakers, businesses, and consumers to make more informed decisions regarding the food we consume.

Sammandrag

Denna uppsats fokuserar på att uppskatta den verkliga kostnaden för matkonsumtion i Sverige genom att bedöma de miljö- och hälsorelaterade externaliteterna kopplade till olika livsmedelskategorier. Den nuvarande livsmedelsmarknaden tar inte hänsyn till dessa dolda kostnader, vilket leder till ohållbara konsumtionsmönster och betydande samhälleliga bördor. Studien använder True Cost Accounting för att kvantifiera och monetarisera olika externaliteter såsom global uppvärmning, övergödning, ekosystemförstöring, förlust av biologisk mångfald, ekotoxicitet, mänsklig toxicitet och sjukdomsbördor. Genom att använda data från databaserna Sustainability Assessment of Foods and Diets och Global Burden of Disease ger forskningen en omfattande översikt av de miljö- och hälsoeffekter som matkonsumtion har i Sverige.

Resultaten visar på en total extern samhällskostnad på 301 miljoner SEK, där 81% härstammar från hälsorelaterade externaliteter och 19% från miljörelaterade. När dessa kostnader beaktas är den verkliga kostnaden för svensk livsmedelskonsumtion nästan dubbelt så hög som den nuvarande nationella livsmedelsutgiften. Externaliteter kopplade till kostrelaterade hälsorisker och koldioxidutsläpp från livsmedelsproduktion är de största bidragande faktorerna. Livsmedelskategorier baserade på kött står för en stor del av både miljöförstöring och hälsorelaterade kostnader, medan andra kategorier såsom fullkornspasta och grönsaker kan ge en samhällsekonomisk nettofördel per kilogram tack vare sina positiva hälsoeffekter. Uppsatsen diskuterar även möjligheterna att använda marknadsbaserade styrmedel, såsom skatter och subventioner, för att internalisera dessa kostnader och främja en mer hållbar och hälsosam livsmedelskonsumtion i Sverige. Genom att inkludera dessa externaliteter i livsmedelspriserna syftar studien till att ge insikter som kan vägleda beslutsfattare, företag och konsumenter mot mer informerade beslut om den mat vi konsumerar.

Keywords: True Cost Accounting, Climate, Food consumption, Externalities, Health, Environment, Sustainability.

Acknowledgements

This thesis project began in January 2025 and was completed in May of the same year. It has been a rewarding and insightful experience, particularly given that the applied assessment method had not previously been used to evaluate the overall external costs of food consumption in a Swedish context. We would like to express our sincere gratitude to all the experts and researchers who supported us throughout this journey.

A special thank you to our supervisors Martin Persson and Jörgen Larsson at the institution of Physical Research Theory at Chalmers University of Technology, for their invaluable support over the course of this project. We are truly grateful for their guidance, critical insights and encouragement, all of which were instrumental in making this report possible.

Gothenburg, May 2025

Glossary of Abbreviations

Below is the list of abbreviations that have been used throughout this thesis listed in alphabetical order:

Abbreviation	**Explanation**
a.i.	Active Ingredient
AMR	Antimicrobial Resistance
CO_2e	Carbon dioxide equivalents
DALY	Disability-Adjusted Life Year
EEA	European Environment Agency
GBD	Global Burden of Disease
GDP	Gross Domestic Product
GHG	Greenhouse Gases
HDI	Human Development Index
IAM	Integrated Assessment Model
LCA	Life-Cycle Assessment
MSA	Mean Species Abundance
QALY	Quality-Adjusted Life Year
SAFAD	Sustainability Assessment of Foods and Diets
SCB	Statistics Sweden (Statistiska Centralbyrån)
SCC	Social Cost of Carbon
TCA	True Cost Accounting
TMREL	Theoretical Minimum Risk Exposure Level
WTP	Willingness To Pay

Contents

List of Figures

List of Tables

1
Introduction

In recent decades, shifts in global food production and consumption have fundamentally transformed the way we eat, with significant implications for both sustainability and public health [1]. As the global food system is increasingly recognised as a major contributor of environmental degradation and disease burden, there is a growing need to reassess how we value food. Rising populations and incomes have led to an overall increase in food demand, particularly for energy-dense products such as high-fat and animal-based foods. The production and consumption of these foods often come at the expense of nutritional quality and environmental resilience, placing significant strain on ecosystems and contributing to greenhouse gas emissions [2], biodiversity loss [3], land degradation [4], excessive pesticide use [5], and high water consumption [6]. In addition to environmental impacts, the modern food system undermines efforts to promote healthy, sustainable diets, exacerbating global health challenges [7].

Sweden is no exception to these trends. Figure 1.1 illustrates the volume of food demand in Sweden between 2018 and projections for 2030 [8]. This growing demand mirrors global patterns, with a shift toward more resource-intensive and less nutritionally beneficial foods.

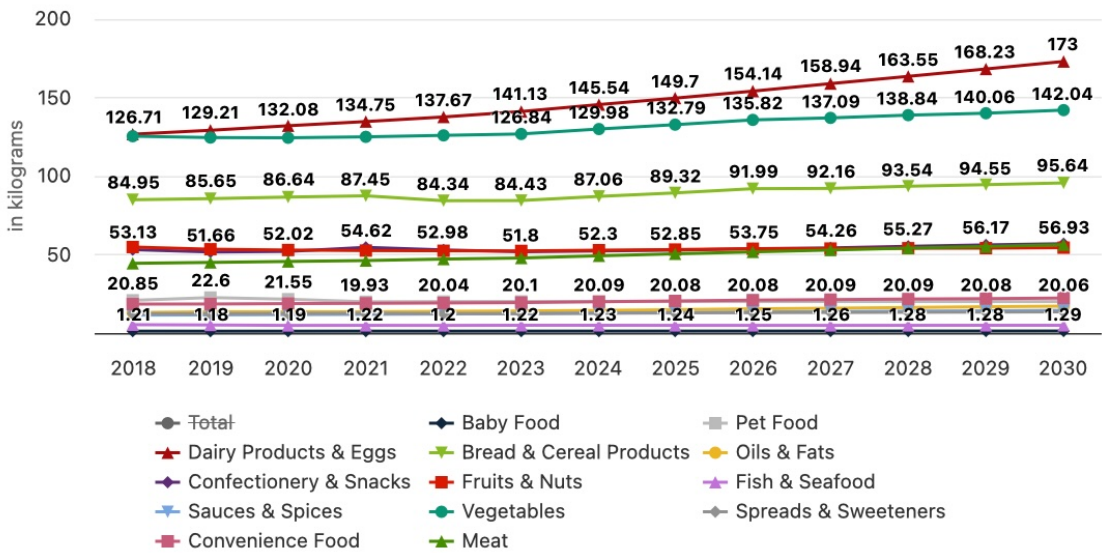

Figure 1.1: Volume of food demand per capita in Sweden from statistics and expectancy from 2018–2030, figure reproduced from Statista in 2025 [8]

In Sweden, the gap between the price consumers pay for food and the broader societal impacts remains substantial, largely due to unaccounted externalities. An externality is a cost or benefit of an activity that affects an uninvolved third party, without being reflected in the market price [9]. This report applies a True Cost Accounting (TCA) approach to explore how environmental and health-related externalities from Swedish food consumption can be quantified and internalised.

1.1 Consequences of Swedish food consumption

In 2022, Swedish agriculture was responsible for emitting 6.4 million tonnes of carbon dioxide equivalents (CO_2e), accounting for approximately 14% of the country's total territorial emissions [2]. However, when emissions are instead calculated based on food consumption, the figure equates to 15 million tonnes of CO_2e [10]. The climate effects of Swedish food consumption occur both within Swedish territory and in other countries through food imports and it is estimated that about 60% of all greenhouse gas emissions from Swedish food consumption occur abroad [11]. Furthermore, Swedish agriculture accounts for approximately 90% of the national ammonia emissions. Along with nitrogen leakage and other pollutants, these emissions contribute to deterioration of air quality, acidification, and eutrophication of water and soil [10].

Similarly, food consumption places an increasing strain on public health. In 2019, dietary habits was identified as the third largest contributor to disease burden in the country [7]. This raises urgent questions about how to balance food security, environmental sustainability, and population health in the years ahead. The poor nutritional content and high energy density of the food we consume pose an increased risk of malnutrition and diseases such as obesity, cardiovascular diseases, metabolic disorders as well as negatively impacting mental health and perceived well-being [12]. The prevalence of obesity or overweight has risen to more than 50% of the population, from Body Mass Index (BMI) measurements [13], following a global trend, while over 2 million people are suffering from cardiovascular diseases [14]. The Swedish Public Health Agency (Folkhälsomyndigheten) emphasises that high intake of sugar, salt, and fats, along with a low intake of fibres and nutritional legumes, are a few of the dietary factors contributing to the large disease burden in Sweden.

An example of this is the intake of red and processed meat, which typically contains high levels of salt and fat and can increase the risk of the diseases previously mentioned. The Swedish Food Agency recommends limiting prepared red meat intake to less than 350 grams of prepared product per week [15]. Precise data on actual intake is unavailable because total consumption is based on import and sales figures, and the reported weight refers to raw meat, which is heavier and often includes bones. However, the Swedish Board of Agriculture could estimate an average prepared red meat consumption of 511 grams per week [16] based on a national dietary survey conducted in 2010–2011 [17]. It is important to note that this average masks substantial variation within the population. While some individuals consume well above the recommended limit, others eat little or no meat. As a result, a certain proportion of the population faces elevated disease risks, even though the national average meat consumption remains below the EU average.

The usage of pesticides, herbicides, and other chemicals and toxins used to enhance crop yields adversely affects both human and animal health [18]. In addition, the excessive use of pharmaceutical products, such as antibiotics in animal production to prevent diseases and loss of livestock, can cause antimicrobial resistance [19]. This constitutes a global health risk that could heavily impact medical treatments and endanger both human and animal life. Although antibiotic usage in Swedish animal production remains relatively low, a portion of imported meat comes from countries such as Denmark, Ireland, the Netherlands, Germany, Poland, and Spain, where antibiotic usage is significantly higher, as illustrated in Figure 1.2.

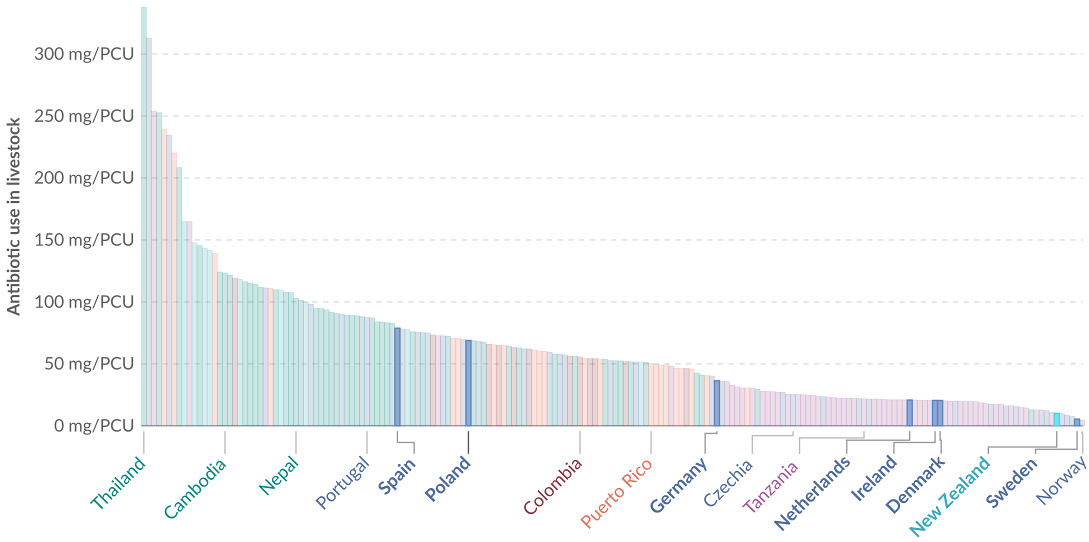

Figure 1.2: Milligrams of total antibiotic use per kilogram of livestock in 2020. Adjusted for differences in livestock numbers and species by standardising to a population-corrected unit (PCU) [19]

1.2 The economic burdens of environmental impact and public health effects

The current agri-food systems have negative impacts both locally and globally, affecting areas such as environmental, health and social well-being [20]. These impacts affect both current and future generations. For example, the current externalised environmental costs are estimated at USD 7 trillion and the costs to human life at USD 11 trillion [21]. In comparison, the total global food consumption is valued at USD 9 trillion.

The negative consequences of Swedish food consumption include considerable health-related costs, as well as expenses linked to work absence, such as sickness benefits and reduced productivity. The societal cost of the 1.35 million Swedes that suffered from obesity in 2022 amounted to SEK 125 billion [2]. Meanwhile, the total cost of cardiovascular diseases was estimated at SEK 60 billion in 2019 [14]. Environmental externalities are challenging to quantify because the estimation of mitigation and adaptation costs does not fully capture the intrinsic value of the loss of environmental quality. As a result, different types of cost analyses are required to estimate these damages accurately. For example, to illustrate the scale of the costs, climate-related adaptation of infrastructure in Sweden was estimated in 2021 price levels to range between SEK 148 and 221 billion. [22].

A fundamental reason for the current unsustainable and unhealthy food system is that market prices fail to reflect the external costs. These externalities include the burden on healthcare, productivity losses, and environmental degradation. Since they are not captured in food prices, these costs are often overlooked by both consumers and policymakers. Utilising a TCA approach enables the identification, quantification, and monetisation of these externalities. This provides greater transparency regarding the real cost of food and supports more informed decisions.

In recent years, there has been an increased focus on health related taxes on food products. Multiple health organisations, such as the World Health Organisation (WHO) and the American Heart Association, recommend a tax on sugar-sweetened beverages as a part of government policy to decrease the risk of diseases such as diabetes and cardiovascular diseases [23]. In 2022, the WHO released a manual for such a tax as a guide for policy makers to promote healthy diets [24]. As of 2022, taxes on at least one type of sugar-sweetened beverage had been applied in at least 108 countries worldwide, Sweden not included, although the coverage of these taxes varies [25].

A notable example is in the UK, where a sugar tax has contributed to the reduction of sugar levels in soft drinks. This was done by introducing a producer fee that increases with the sugar content [26]. Another case where market based interventions have been implemented was in Denmark, where a tax was introduced on saturated fats above a certain threshold [27].

1.3 Previous research

Despite TCA being a fairly new approach, several studies have explored its application in the food system, highlighting the societal costs of food production and consumption. Two such reports by Perotti [28] in Switzerland and The Rockefeller Foundation in the US [29] conducted TCA assessments, quantifying the externalities and their impacts. Their findings demonstrated significant differences in cost between food categories.

Various policy instruments, such as environmental taxes and fiscal proposals, have been explored to encourage healthier and more sustainable food choices in Sweden.

One such proposal is the report *Matskatteväxling* by Larsson et al. [30], exploring how altered pricing, based on environmental taxation and subsidies, could incentivise such shifts in food consumption.

1.4 Aim and research questions

The aim of this thesis is to estimate the true cost of Swedish food consumption by monetising key environmental, health-related externalities as well as animal welfare impact. The report applies a TCA approach inspired by previous assessments from other countries, using Swedish data on agriculture, food imports, and consumption patterns. This thesis will also quantify and compare the external costs of the main food categories, such as meat, dairy, vegetables, and fruit, and examine how these costs differ. The findings of the thesis are intended to serve as a tool for guiding market interventions, such as taxes or subsidies, that support a shift toward a more sustainable and health-oriented food consumption in Sweden.

Since estimations of external costs often involve significant underlying uncertainties and methodological differences, valuations may vary significantly between sources and are often presented as a range (e.g., low, base, and high estimates). Therefore, a sensitivity analysis will be conducted to assess how the choice of valuation influences the results.

The study addresses the following research questions:

- *What is the true cost of Swedish food consumption, considering environmental and health-related externalities, as well as animal welfare?*
- *How do the external costs of different food categories compare?*
- *How does the choice of valuation method influence the estimated cost of Swedish food consumption?*

1.5 Scope

This study assesses the true cost of food consumption in Sweden, primarily focusing on environmental and health-related externalities. Although numerous externalities could theoretically be included, the analysis prioritises those that are measurable, quantifiable, and supported by available data. Other potential externalities, such as those related to socioeconomic disparities and labour conditions in food production, are excluded due to data limitations and methodological constraints. Among the social externalities, only animal welfare is included, since reliable data is available. The selection of externalities is further constrained to those that are both relevant to the Swedish context and supported by sufficient data. In cases where Sweden-specific data is unavailable, international sources are used.

The analysis is conducted for both the total annual external costs of Swedish food consumption and the cost per kilogram for aggregated food categories. Within each category, the externality values reflect weighted averages across multiple products. As a result, differences between individual products, brands, and production methods, such as conventional and organic farming, are not explicitly captured in the analysis.

2
Theory

This chapter outlines the theoretical foundations and key concepts relevant to assessing the true cost of food consumption in Sweden, and aims to lay a foundation for understanding the need for market-based interventions for more sustainable consumption patterns.

2.1 Externalities

Externalities represent a form of market failure as they lead to an inefficient allocation of resources [31]. Since these external effects are not reflected in the market price, overproduction and overconsumption of some products are possible consequences. When this leads to pressure on natural resources, it is a negative externality, which means that the production of goods leads to costs for society. There are also examples of positive externalities that benefit society, such as the consumption of certain products contributing to better public health. These externalities can lead to underproduction and underconsumption of a product, as their broader societal values are not reflected in the market price [32].

The United Nations food summit 2021, with the aim to develop strategies to achieve the UN's 17 sustainable development goals, addressed the issue of externalised costs in the food system [21]. It emphasises how unsustainable, unaffordable and unhealthy food production and consumption arises because of externalities making these types of food more profitable to produce. As a result, healthier and more sustainable food options are often more expensive for consumers and less profitable for businesses. This market failure arises from an imbalance, as the actors who profit from the food system are not the ones bearing the full costs it imposes on society and the environment. To address these externalities and to transition the food system toward greater sustainability and health, many economists agree on the need to internalise the externalities of food products in their prices through true pricing [21], [31].

2.2 True Cost Accounting

Although the need to internalise externalities has been acknowledged for more than a century, there have been difficulties in quantifying and valuing them [31]. The internalisation of externalities refers to the process of including the total costs and benefits in the pricing of goods and services. This can involve measures to reduce or prevent negative externalities or create positive externalities, and can be done either through business models or regulatory policies. TCA is a relatively new alternative that provides frameworks for identifying, quantifying and valuing externalities in order to internalise them [21]. The method utilises a holistic approach used to assess the societal costs related to production and consumption [33]. In contrast to traditional economic accounting, TCA frameworks provide a more comprehensive understanding of the actual impact.

Quantifying externalities can be done with a variety of methods, one common technique being Life Cycle Assessment (LCA) [33]. LCA is a standardised methodology used to evaluate the environmental and social impacts of a product throughout its entire value chain [34]. This framework enhances transparency by quantifying emissions and resource use, identifying critical areas where interventions can reduce negative effects on ecosystems and human health. To ensure consistency and credibility, these analyses follow international guidelines such as the ISO 14040 and 14044 standards, which are described in the International Reference Life Cycle Data System developed by the European Commission.

While the results of an LCA is typically expressed in natural units, i.e. the units the externalities were originally measured in, many quantitative TCA frameworks translate the environmental and social impact into monetary units [33]. Monetary valuation can be done with a variety of means, depending on the nature of the externality and data availability [32]. These valuation techniques include using market prices and other empirical methods such as:

- Production factor: Measuring how an externality affects production, e.g. crop yield or forest growth.
- Contingent valuation: Measuring the social costs of an externality through surveys.
- Averting expenditures: Estimating environmental externalities based on the costs of avoiding negative environmental impacts, e.g. the cost of installing water filters to reduce the amount of nitrates in the drinking water.

A key concept in environmental economics is an individual's willingness to pay (WTP) for a certain service or good [32]. It is often assessed through the contingent valuation method, which can be used to estimate the value that individuals place on non-market goods and services such as cultural, environmental, and health issues [35]. Contingent valuation lets economists simulate a market by giving respondents a hypothetical scenario and asking them how much they are willing to pay for a specified level of a good, or a change in its quality [36]. The method provides a relatively inexpensive way of collecting data directly from consumers instead of relying on secondary data. However, this comes with a few uncertainties. Since they are not faced with a real financial commitment, respondents might over- or underestimate their actual willingness to pay in a certain scenario. Lack of information, as well as other kinds of bias, can also affect the reliability of the results.

TCA frameworks are particularly useful in the food and agriculture sector in order to measure and value both the positive and negative environmental, health and social costs [33]. The increased use of TCA has led to the development of numerous methodologies. Although this has contributed to greater transparency in assessing the true costs of food systems, the diversity of approaches has also made it more challenging to directly compare results across the different frameworks. Within the frameworks, a variation of system boundaries, units, and monetisation methods creates inconsistency, consequently limiting qualitative comparisons of the outcomes.

3

Methods and materials

This chapter describes the methodology used to quantify the externalities associated with food consumption in Sweden. Both environmental and health related external costs have been considered and translated into monetary values. The methods cover national level estimates as well as pricing for different food categories where data availability allowed for further breakdown.

3.1 Affected areas & assessment of externalities

A range of potential externalities were initially identified to assess the impacts of Swedish food consumption. This early selection was guided both by the relevance of the indicators and the expected availability and quality of the data. The aim was to establish a comprehensive overview of possible areas of impact before narrowing the scope for the final quantitative analysis. For clarity and comparability, the externalities were grouped into three major impact categories: environmental, health and social. This categorisation allows for a clearer understanding of how different types of costs are distributed and enables comparison between them. Among these, animal welfare is the only social externality considered. It is therefore treated as a distinct category due to its ethical dimension, separate from health and environmental valuations.

Table 3.1: Considered environmental, social and health externalities, categorised by indicator

Indicator	Externality	Description
Carbon footprint	Climate change	Emissions of greenhouse gases contributing to global warming
New N input	Eutrophication	Nutrient overload in water bodies leading to algal blooms, oxygen depletion, and loss of aquatic biodiversity
New P input	Eutrophication	Nutrient overload in water bodies leading to algal blooms, oxygen depletion, and loss of aquatic biodiversity
Blue water use	Water scarcity	Excessive freshwater withdrawal from groundwater and surface water bodies like rivers and lakes
Pesticide use	Ecosystem toxicity	Chemical exposure harming biodiversity and ecosystems
Ammonia emissions	Eutrophication	Airborne nitrogen contributing to nutrient overload, leading to algal blooms, oxygen depletion, and loss of aquatic biodiversity
Cropland use	Biodiversity loss and habitat conversion	Occupation of land for agriculture preventing other ecological functions, displacing natural habitats and ecosystems
Animal welfare index	Animal suffering	Impact on animal health and well-being
Dietary risks	Health impact of diet related diseases	Disease burden from under- or overconsumption of foods and nutrients
Pesticide use	Human toxicity	Health effects from occupational exposure, local environmental exposure, and dietary residues
Air pollution	Human toxicity	Disease burden due to decreased air quality
Antibiotic use	Health impact due to antimicrobial resistance	Health burden due to reduced antibiotic efficacy and increased treatment failures
Heavy metal exposure	Human toxicity	Toxicity from foodborne heavy metals

Table 3.1 presents the range of externalities initially considered, with environmental, social, and health-related impacts highlighted in green, blue, and pink, respectively. Not all of these were included in the final calculations, as some lacked sufficient data or valuation methodologies.

3.2 Data collection

To assess the externalities related to food consumption, the study relies on data from multiple sources covering both environmental and health impacts. Two main databases are used: the Sustainability Assessment of Foods and Diets (SAFAD), which provides data on sustainability footprints, and the Global Burden of Disease database (GBD), which offers estimates of health impacts linked to various risk factors, including dietary risks.

3.2.1 Sustainability Assessment of Foods and Diets

One of the main sources for data collection on externalities is the SAFAD database, an open access tool for assessing the impact of foods and diets [37]. The tool includes eight environmental indicators along with indices for animal welfare and antibiotic use. The footprint of each product is representative of the Swedish market, as calculations are based on import shares for the raw commodities and the amount required of each related ingredient [38]. The impact of each product is calculated throughout the whole value chain and includes emissions from raw primary commodities, transportation of the commodities, processing, packaging, and waste in production, retail, and consumption. When available, official statistics from e.g. the Food and Agriculture Organization Statistics (FAOSTAT) and the European Statistical Office (Eurostat), as well as data from trade organisations and scientific literature such as published LCAs, were used in the calculations. In cases where there were gaps in the available data, extrapolation and approximations were applied. The resulting impacts are presented per kg of product for each food item.

To connect external costs to specific foods, this study uses food categories rather than individual food items. This approach simplifies the analysis while still reflecting consumption patterns and impact. The categorisation is based on 38 food groups developed by Larsson et al. [30], which is based on data from the SAFAD database. Each category represents a weighted average of the 20 most sold food products by weight within that group, ensuring that the values reflect real consumption behaviour. These categories collectively cover the entire food consumption in Sweden, making them suitable for national-level estimations.

The food categories developed by Larsson et al. [30] are used throughout the study as a common structure to organise and present the externalities. The SAFAD dataset contains relevant data linked to these categories, particularly for aspects related to production. Additional sources were used to complement this information, especially for health-related impacts where SAFAD did not provide sufficient coverage. Wherever possible, externalities were matched to the same category structure to ensure coherence across the analysis. The footprints from SAFAD used in this report can be seen in Table 3.2.

Table 3.2: Sustainability footprints and corresponding units, based on SAFAD [37]

Footprint Category	**Unit**
Carbon footprint	kg CO_2e
New N input	kg N
New P input	kg P
Blue water use	m^3
Pesticide use	g a.i.
Ammonia emissions	kg NH_3
Cropland use	$m^2 \cdot$ year
Animal welfare	Index
Antibiotic use	Index

3.2.2 Global Burden of Disease

The GBD study is the most extensive and detailed scientific initiative developed to quantify health trends and risks [39]. Led by the Institute for Health Metrics and Evaluation (IHME) at the University of Washington, and supported by a global network of over 12,000 researchers, the GBD framework provides comprehensive estimates of disease burden and risk factors across 204 countries. GBD combines diverse data sources to produce standardised estimates of disease burden, allowing for comparisons across time, geography, and health conditions.

A key metric developed by the GBD study is the Disability-Adjusted Life Year (DALY), which captures the total burden of disease by combining the years lost due to premature mortality (YLL) and years lived with disability (YLD) [40]. This relationship can be expressed as:

$$\mathrm{DALY} = \mathrm{YLL} + \mathrm{YLD} \tag{3.1}$$

One DALY represents one year of healthy life lost. This metric is particularly useful in identifying diseases that may not be fatal but still impose a significant burden through long-term disability.

Quality-Adjusted Life Year (QALY), on the other hand, is more commonly used in health economics and clinical decision making [41]. It reflects both the quality and the quantity of life lived. One QALY equates to one year in perfect health. If a person lives for a year in a health state valued at 0.5 (on a scale from 0 = death to 1 = perfect health), this would be equivalent to 0.5 QALYs.

While both DALY and QALY aim to quantify the burden of disease, they differ in perspective: DALYs focus on lost health and are often used to assess population-level health burdens [40], whereas QALYs emphasise gained health and are primarily applied in evaluating healthcare interventions at the individual level [41].

To attribute disease burden to specific risk factors, such as diet or air pollution, GBD applies a comparative risk assessment framework [42]. This method estimates the theoretical reduction in disease burden if the population were exposed to an optimal exposure level, called the Theoretical Minimum Risk Exposure Level (TMREL). Risk relationships are modelled using established relative risks from meta-analyses, applied across population distributions stratified by age, sex, and geography. Non-linear dose–response functions are often used, meaning the health risk does not necessarily increase proportionally with exposure. Importantly for this study, the GBD estimates used are those specifically calculated for Swedish population characteristics, including national patterns of dietary intake, disease rates, and demographic profiles.

Table 3.3: Dietary risk factors, associated disease outcomes, and the optimal intake levels of foods associated with risk (TMREL), based on GBD 2021 [42]

Dietary Risk Factor	Associated Diseases	TMREL
Diet low in fruits	Cardiovascular diseases, type 2 diabetes, chronic kidney disease, tuberculosis, respiratory cancers	340–350 g/day
Diet low in vegetables	Cardiovascular diseases, type 2 diabetes, chronic kidney disease, tuberculosis, esophageal cancer	306–372 g/day
Diet low in legumes	Ischemic heart disease	100–110 g/day
Diet low in whole grains	Cardiovascular diseases, type 2 diabetes, chronic kidney disease, tuberculosis, colon and rectum cancer	160–210 g/day
Diet low in nuts & seeds	Ischemic heart disease	19–24 g/day
Diet low in milk	Prostate cancer, colon and rectum cancer	280-340 g/day (m)[a] 500-610 g/day (f)[a]
Diet high in red meat	Cardiovascular diseases, type 2 diabetes, chronic kidney disease, tuberculosis, colon and rectum cancer	0–200 g/day
Diet high in processed meat	Cardiovascular diseases, type 2 diabetes, chronic kidney disease, tuberculosis, colon and rectum cancer	0 g/day
Diet high in sugar-sweetened beverages	Cardiovascular diseases, type 2 diabetes, chronic kidney disease, tuberculosis	0 g/day
Diet low in fiber	Cardiovascular diseases, type 2 diabetes, colon and rectum cancer	22–25 g/day
Diet low in calcium	Prostate cancer, colon and rectum cancer	0.72–0.86 g/day (m)[a] 1.1–1.2 g/day (f)[a]
Diet low in seafood omega-3	Ischemic heart disease	470–660 mg/day
Diet low in polyunsaturated fatty acids	Ischemic heart disease	9–10% of total daily energy
Diet high in trans fatty acids	Ischemic heart disease	0–1.1% of total daily energy
Diet high in sodium	Cardiovascular diseases, chronic kidney disease, stomach cancer	1–5 g/day

[a] m = males, f = females

Table 3.3 illustrates the links between various dietary risk factors and disease outcomes. However, most dietary risk factors do not directly cause diseases, but instead exert their effects through intermediate biological mediators such as elevated blood pressure, high blood glucose, or low nutrient levels [42]. This structure is reflected in the GBD 2021 model, where associations are mapped through these mediators before leading to specific health outcomes. For example, a diet low in milk increases the risk of colon and rectum cancer primarily by contributing to low calcium intake, while high sodium intake elevates the risk of cardiovascular diseases through its impact on systolic blood pressure.

3.3 Assumptions and methodology limitations

Throughout this study, several assumptions and limitations were made to enable analysis despite constraints in data availability, accuracy, and scope. These were necessary to ensure feasibility but also to introduce uncertainties that should be considered throughout the report.

3.3.1 Data consistency

It is assumed that the data sources used are comprehensive and sufficient representations of real-world conditions. However, potential quality gaps can affect the accuracy of the results. Variations and missing data could introduce bias, which leads to sensitive estimations of the findings. Moreover, data from different sources may have been composed using different assumptions and limitations.

3.3.2 Extrapolation and generalisation

In cases where specific data was unavailable, values from similar studies or contexts were used as proxies. This allowed for broader extrapolation, but also introduced uncertainty, as some estimates may not fully reflect Swedish conditions. While some data rely on national or regional averages, local variations in environmental and health factors may be overlooked. As a result, applying these values across different contexts may affect the accuracy of the findings.

3.3.3 Avoidance of double counting

Efforts were made to prevent overlap in impact calculations. However, due to interconnected factors and unclear descriptions for some data sources, there remains a risk that certain parts have been counted twice. This could alter the final results, potentially leading to an overestimation of the true impacts.

3.3.4 Linearity

Parts of the methodology is based on the assumption of linear relationships between input variables and their associated impacts. This means that changes in one factor are assumed to result in proportionally equal changes in outcomes. While this simplification facilitates analysis and comparability, it may not fully capture complex, non-linear dynamics present in real-world systems.

3.3.5 Temporal variations

Variations in time could introduce uncertainty of the results. Data used for monetisation terms and production numbers was collected over varying years. This temporal variation can lead do reduced consistency and weaker estimations. Moreover, food production systems and dietary patterns are dynamic and constantly evolving, consequently failing to accurately capture future trends or shifts in consumer behaviour. Additionally, since some of the externalities accumulate progressively over time, it complicates the estimate predictions over time. Annual estimates may fail to fully represent the long-term impacts for some of these externalities.

3.3.6 Categorisation of food groups

In this study, food items have been categorised into broader groups such as processed meat and vegetables rather than the individual products. While this has simplified the calculation process, it may also mask significant variations in environmental and health impacts within those groups. This is not only relevant for the individual products itself, but also that different production methods may have been used. The distinction between organic and conventional production becomes less apparent once the products are categorised.

3.4 Environmental impacts

The assessment of environmental externalities is primarily based on a meta-analytical approach, drawing on comparisons across existing literature. Using the environmental footprint categories from the SAFAD database as a foundation, each category's impact was examined and corresponding monetary values were assigned where possible. Certain categories were excluded due to data limitations and high levels of uncertainty. A summary can be seen in Table 3.4. The methodology for each category will be further explained in the following sections.

Table 3.4: Methods for economic valuation of environmental externalities by indicators for Swedish food consumption

Indicator	**Calculation**
Carbon footprint	Estimated global damage costs using the SCC [43], applied to CO2e emissions
New N input	WTP-based estimate using regional valuation of nitrogen reduction to the Baltic Sea [44], adjusted for excess nutrient use
New P input	Excluded
Blue water use	Excluded
Pesticide use: Ecotoxicity	Estimated using the PEA model, GDP-adjusted for Sweden based on data from the UK, the US, and Germany covering cost of remediation of damaged habitats[45].
Ammonia emissions	A WTP-based estimate using regional valuation of nitrogen reduction to the Baltic Sea, adapted and scaled for Swedish ammonia emissions [44]
Cropland use	Compensation costs based on the value of ecosystem services lost due to land use [46]

3.4.1 Carbon footprint

Carbon footprints are typically measured in carbon dioxide equivalents (CO_2e), a standardised metric that expresses the global warming potential (GWP) of various greenhouse gases (GHG) relative to the impact of 1 kilogram of carbon dioxide (CO_2) [47]. This allows emissions of gases like methane or nitrous oxide to be compared and aggregated in a common unit.

To assign a monetary value to carbon emissions, two primary approaches are commonly used: the Social Cost of Carbon (SCC) and the Marginal Abatement Cost (MAC). A practical baseline for MAC values can be seen in carbon pricing mechanisms like the European Union Emissions Trading System (EU ETS) [48]. The EU ETS operates as a cap and trade system where emitters must purchase allowances for their emissions, effectively creating a market price for carbon. In this case, the MAC represents the cost of reducing one additional metric ton of CO_2 emissions and are therefore reflecting market based mitigation costs [49].

SCC, on the other hand, refers to the estimated economic damage resulting from emitting one additional metric of CO_2 into the atmosphere [43]. It's a policy driven metric capturing a wide range of external costs globally using Integrated Assessment Models (IAMs). The SCC is highly sensitive to the discount rate, which reflects how we value future costs and benefits relative to those in the present. The discount rate determines how much weight we place on the future damages caused by GHG emissions today. A higher discount rate means that we place less value on future damages, making today's emissions seem less costly in economic terms. This results in a lower SCC. The discount rate is highly political because it reflects ethical judgments about future interests to present economic priorities. Governments and institutions therefore debate what rate is appropriate. For example, the US government has used discount rates ranging from 1.5% to 2.5% in estimating the SCC.

Table 3.1 presents the estimated social costs for three major GHGs: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) [43]. These costs are projected from 2020 to 2080, under different discount rate scenarios. The wide variation in cost across gases reflects their differing GWPs and atmospheric lifetimes, which translate into distinct levels of environmental harm per unit emitted. In this analysis, the values for the year 2020 are used, based on the availability and reliability of published estimates as well as the intention to align the valuation with current economic conditions. However, future SCC estimates are expected to project higher costs as a results of increasing climate impacts [43].

Table ES.1: Estimates of the Social Cost of Greenhouse Gases (SC-GHG), 2020-2080 (2020 dollars)

	SC-GHG and Near-term Ramsey Discount Rate								
	SC-CO_2 *(2020 dollars per metric ton of CO_2)*			SC-CH_4 *(2020 dollars per metric ton of CH_4)*			SC-N_2O *(2020 dollars per metric ton of N_2O)*		
Emission Year	2.5%	2.0%	1.5%	2.5%	2.0%	1.5%	2.5%	2.0%	1.5%
2020	120	190	340	1,300	1,600	2,300	35,000	54,000	87,000
2030	140	230	380	1,900	2,400	3,200	45,000	66,000	100,000
2040	170	270	430	2,700	3,300	4,200	55,000	79,000	120,000
2050	200	310	480	3,500	4,200	5,300	66,000	93,000	140,000
2060	230	350	530	4,300	5,100	6,300	76,000	110,000	150,000
2070	260	380	570	5,000	5,900	7,200	85,000	120,000	170,000
2080	280	410	600	5,800	6,800	8,200	95,000	130,000	180,000

Figure 3.1: Social cost of CO_2, CH_4 and N_2O (from left to right) calculated from Climate Impact Model developed by the Climate Impact Lab (CIL, Carleton et al., Rode et al.)), Greenhouse Gas Impact Value Estimator (GIVE) model [50] and Meta analysis global damage function estimation [51] over time.

In Sweden, a fixed carbon price has been set by policymakers as a tool to reduce emissions by influencing consumer and producer behaviour [52]. Through the implementation of carbon taxes, the cost of emitting CO_2 is internalised into the price of fossil fuels, effectively reducing the purchasing power for carbon intensive goods and services. The tax rates vary depending on the type of fuel, reflecting differences in carbon content.

The tax per ton of CO_2 is derived from a conversion formula with standardised emission factors [53], and corresponding fuel tax [52]. For example, for petrol:

$$\text{Carbon tax per kg } CO_2 = \frac{3.14}{2.36} = 1.33 \text{ SEK/kg } CO_2 \tag{3.2}$$

MAC pricing is based on WTP, as determined through the auction mechanism of the EU ETS. Accordingly, it is presented in Table 3.5 using an average value, as well as high and low auction price level from 2024. The SCC is shown with corresponding discount rates of 1.5%, 2%, and 2.5%, representing a range of high, medium, and low price respectively. Lastly, the carbon tax price is fixed per kilogram of CO_2 and presented in Table 3.5 for 2024 medium price rate.

Table 3.5: Carbon pricing in 2024 price level

Approach	Scope	Price (SEK/kg)			Source
		Avg/mid	High	Low	
Policy	Carbon taxes in Sweden	1.3 *	-	-	Swedish Tax Agency, 2024 [52]
MAC	Auction price EU ETS	0.74	0.86	0.57	EEX, 2025 [54]
SCC	Estimation of damage costs for CO_2	2.2	3.9	1.4	EPA, 2023 [43]

* Carbon tax rate for petrol (motor gasoline) with environmental class 1 (miljöklass 1)

Although SCC and MAC originate from different frameworks, they can converge under optimal climate policy conditions. In theory, if a government sets a carbon price equal to the SCC, then market actors would reduce emissions up to the point where the MAC of the last unit reduced equals the SCC [43]. In practice, however, most existing carbon prices fall short of the estimated SCC. For example, although Sweden's carbon tax is relatively high compared to the MAC implied by the EU ETS, it remains significantly below the SCC estimates derived from scientific assessments of global climate damages.

In this assessment, the SCC is applied as the valuation metric for carbon emissions, as it better reflects the full societal cost of climate related damages. Using a lower price, such as the existing carbon tax, would risk underestimating the true external cost of emissions and preserve the gap between market signals and actual environmental harm.

3.4.2 New N input and new P input

The new nitrogen (N) and phosphorus (P) input categories from SAFAD quantify the added nutrients from mineral fertilisers and biological fixation [38]. However, these indicators do not account for the impact on terrestrial or aquatic ecosystems, since the effects are highly dependent on local conditions, such as soil type, topography, climate, crop type, and the chemical form of the nutrient [55]. Since these factors are difficult to assess, a simplified approach was used, as follows:

$$\text{Nutrient input to the production system} - \text{Nutrient output via harvested products} \\ = \text{Excess nutrients} \tag{3.3}$$

It was assumed that all excess nutrients have the potential to contribute to eutrophication. To get the share of excess nutrient use compared to total nutrient input, the following formula was used:

$$\text{Share of excess nutrients of total input} = \frac{\text{Excess nutrients}}{\text{Nutrient input to the production system}} \tag{3.4}$$

This was calculated for both nitrogen and phosphorus using data on Swedish agriculture from Statistics Sweden (Statistiska centralbyrån, SCB) [56]. The excess was assumed to be consistent across food categories, resulting in a single average value for each nutrient. The resulting excess nutrients per year for phosphorus was zero. Given this rough estimate, the associated cost of eutrophication is therefore negligible. As a result, a decision was made not to include the cost of phosphorus in the assessment.

For nitrogen, Equation 3.4 yielded a value of approximately 29%. This percentage was then applied to the original SAFAD data for new nitrogen input to estimate the amount of excess nitrogen use.

The cost of eutrophication was based on the Swedish Environmental Protection Agency's (Naturvårdsverket) price database which provides an estimated price for reduced nitrogen emissions to the Baltic Sea [44]. The price was based on the total WTP of all nine countries bordering the Baltic Sea to reduce eutrophication and achieve the goals of the Baltic Sea Action Plan, divided by their combined nitrogen reduction commitments. However, since the emissions do not need to be reduced to zero to maintain good environmental status, applying this price directly to the emissions would overestimate the environmental impact. Therefore, an adjustment of the price was necessary to reflect the cost per kilogram of nitrogen emissions.

To estimate the cost per kilogram of nitrogen emissions, the WTP was instead divided by the total nitrogen emissions, using values obtained from the Baltic Marine Environment Protection Commission (HELCOM) [57]. The inflation-adjusted WTP of SEK 43 billion per year and the total nitrogen input of 977 kilotonnes per year were used, resulting in the following estimate:

$$\text{Price/kg N emissions} = \frac{\text{WTP}}{\text{Total N emissions}} \approx 44.28\,\text{SEK/kg N emissions} \tag{3.5}$$

The cost of food consumption was then calculated as:

$$\text{Cost} = \text{Price/kg N emissions} \times (\text{share of excess nutrient input}) \times \text{New N input} \tag{3.6}$$

3.4.3 Ammonia emissions

Ammonia emissions included in the SAFAD data originate from fertiliser and manure use [38]. While ammonia can contribute to several environmental issues, such as acidification, the price estimate will consider only its effects related to eutrophication.

The Swedish Environmental Protection Agency has estimated the price of reduced ammonia (NH_3) emissions by calculating the proportion of national emissions that reach the Baltic Sea [44]. The deposition of nitrogen compounds in lower oxidation states, such as ammonia, ammonium (NH_4^+), and organic nitrogen, to the Baltic Sea was divided by the total Swedish ammonia emissions. This resulted in a fraction of 0.153. This fraction was then multiplied by their nitrogen reduction price, resulting in the estimated price for reduced emissions of ammonia.

However, as the objective in this report is to determine the cost per kilogram of ammonia emitted rather than reduced, the fraction was instead multiplied by the cost of nitrogen emissions, as estimated in Equation 3.5. This resulted in the following calculation:

$$\text{Price/kg NH}_3\text{ emissions} = 44.28 \times 0.153 \quad \approx 6.77\text{ SEK/kg NH}_3\text{emissions} \tag{3.7}$$

3.4.4 Blue water use

Data provided by SAFAD presents the amount of freshwater used during production [38]. To estimate the cost, it is necessary to determine what portion of this water is scarce, as the environmental impacts depend on it. A restoration cost for scarce blue water use from True Price Foundation, which was set at 1.330 EUR/m^3, was studied [46]. However, it is difficult to determine the exact details required, such as the precise location and local conditions at the time of production. Therefore, this category was excluded, as a reliable estimate could not be made.

Water scarcity is generally low in Sweden with a Water Exploitation Index plus (WEI+) of 0.4%, which can be seen in Figure 3.2. The WEI+ is the percentage of available renewable freshwater resources that is being consumed during the worst quarterly water scarcity conditions of Q3 from July to September [58]. This suggests that domestic water-related environmental costs are minimal. However, Sweden imports a significant share of its food from countries with higher water stress. The largest sources of food imports include the Netherlands with 15% of total imports, Denmark with 14% and Germany with 13%, all of which report higher WEI+ values of 6.0%, 18.6%, and 4.5% respectively, with data from 2024 [59], which can be seen in Figure 3.2. These figures indicate that imported food may be associated with higher levels of water exploitation.

As a result, the exclusion of blue water costs may lead to an underestimation of the total environmental burden, particularly for food categories with high water footprints sourced from water-stressed regions. If these costs were to be fully accounted for, it is likely that significant differences between food categories would emerge in terms of their water-related externalities.

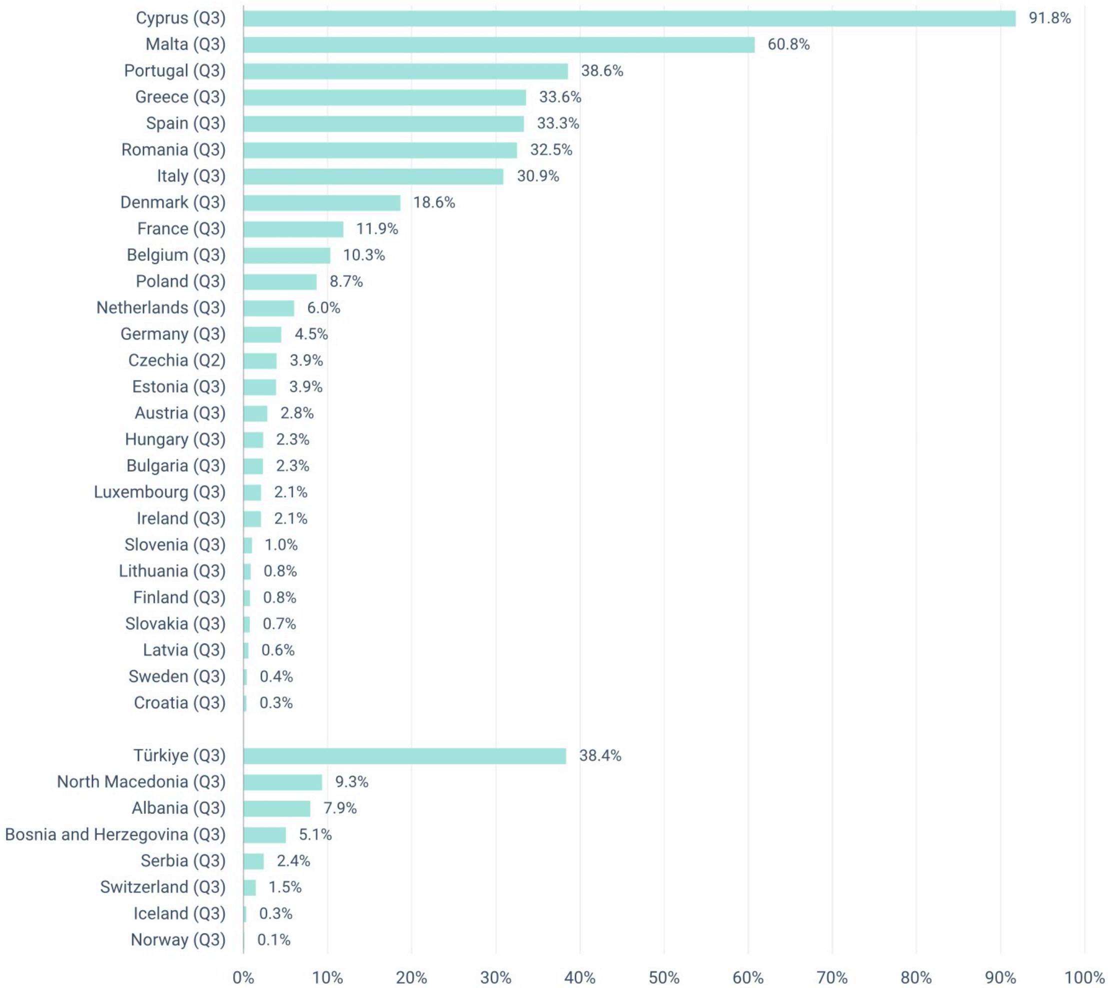

Figure 3.2: Worst seasonal water scarcity conditions (July-September) for European countries in 2022, measured for the WEI+ [58] with data from Joint Research Center (JRC), European Environment Agency (EEA) and Eurostat

3.4.5 Pesticide use: Ecotoxicity

Pesticides such as herbicides, insecticides, nematicides, and fungicides are commonly used in agriculture to improve efficiency and increase yields by targeting fungi, harmful bacteria and insect pests [60]. While effective in boosting production, pesticide use can have substantial negative impacts on ecosystems. This report employs the Pesticide Environmental Accounting (PEA) model to estimate the external environmental costs associated with pesticide applications [45]. The model integrates the Environmental Impact Quotient (EIQ), which quantifies the relative ecotoxicological risk of various pesticide active ingredients (a.i.). EIQ scores are based on toxicity data and environmental persistence, and they are used to weight pesticide impacts by active substance and application rate.

Within the ecological scope of the model, two primary categories of environmental effects are assessed: aquatic and terrestrial [45]. Aquatic effects include pesticide runoff and toxicity to aquatic life such as fish, while terrestrial effects include harm to birds, bees, beneficial insects and chemical persistence in soils and plants.

The external costs incorporated in the model are derived from studies conducted in the United Kingdom (UK), United States (US) and Germany (DE) [45]. To be able to apply the model to Sweden (SE) which is different in terms of both pesticide usage intensity and Gross Domestic Product (GDP) per capita, it is necessary to adjust the cost estimates accordingly. The adjustment uses a GDP-weighted transformation as follows:

$$C_{\mathrm{SE}} = \left(\frac{C_{\mathrm{UK}}}{\mathrm{GDP}_{\mathrm{UK}}} + \frac{C_{\mathrm{US}}}{\mathrm{GDP}_{\mathrm{US}}} + \frac{C_{\mathrm{DE}}}{\mathrm{GDP}_{\mathrm{DE}}} \right) \times \frac{1}{3} \times \mathrm{GDP}_{\mathrm{SE}} \tag{3.8}$$
$$\text{where } C = \text{Mean cost}$$

The PEA model is based on data from three high-income countries where the costs of environmental remediation and monitoring are relatively high. Consequently, the model may overestimate the external costs for countries with lower service costs and weaker enforcement systems and potentially underestimate them, since the costs are based on a WTP.

Furthermore, the extent of environmental damage from pesticides depends not only on toxicity but also on usage intensity, landscape characteristics and ecosystem sensitivity [45]. Although these country specific variables are not directly included in the model, the PEA allows for indirect adjustment by scaling costs according to GDP per capita, which serve as proxies for remediation capacity and population exposure, respectively.

Table 3.6: Estimated external costs of pesticides in SEK (2024 price level), derived from Leach & Mumford [45]

Ecological category	**Mean cost per kg a.i. (SEK)**		
	UK	**US**	**DE**
Pollution incidents, fish deaths and monitoring costs	9.19	3.9	19.9
Biodiversity/wildlife losses	13.7	5.0	2.4
Bee colony losses	1.08	3.55	0.54

Using this method, the estimated average external environmental cost for pesticide use in Sweden is calculated to be approximately 19.5 SEK/kg a.i. This is in comparison with the highest country specific cost, from the UK at 23.9 SEK/kg a.i., and the lowest, from the US at 12.4 SEK/kg a.i. [45]. The adjustment ensures the model reflects Sweden's economic context and agricultural profile more accurately.

3.4.6 Cropland use

The cropland use metric from SAFAD quantifies the land area (m^2) required annually to produce food, as defined by Röös et al. [38]. Cropland use has a significant impact on ecosystem services and biodiversity. The occupation of land for agricultural activities limits its availability for other ecological functions, leading to the displacement of natural habitats and ecosystems. This results in losses in both biodiversity and ecosystem services [61].

The True Price Foundation assigns monetary values to ecosystem services lost due to land use for different biome types, based on a study by De Groot et al. [62]. These values represent compensation costs, which are the estimated value of the ecosystem services no longer provided by the land when it is used for agriculture [46].

To apply these values, specific biome cover data is required to apply the appropriate valuation categories. A biome refers to a large ecological unit defined by dominant vegetation in terrestrial regions or biogeochemical properties in marine areas, within which ecosystems function in broadly similar ways [61]. Since approximately 70% of Sweden is covered by forest, and detailed land cover data is limited, forest was assumed to be the dominant biome nationwide [63]. Therefore, the *Other forest* biome category from True Price was used to represent Sweden's costs due to cropland use.

The data provided by SAFAD is expressed in $m^2 \cdot$ years. In order to apply the True Price monetisation value, this was converted to MSA $\cdot$ ha $\cdot$ years. Mean Species Abundance (MSA) is the proportion of species populations remaining under the current land use compared to its original, undisturbed natural state [61]. An MSA-value of 0.6 for Sweden, sourced from the Global biodiversity model for policy support, was used [64]. The result was then divided by 10,000 to convert square metres into hectares.

All other biome types presented by the True Price Foundation have higher costs than the *other forest* category. Inland wetlands, which exist in Sweden, are valued at more than ten times higher [46]. This suggests that the estimation may be underestimated due to the absence of detailed biome cover data for Sweden. Furthermore, the impact of imported food is not accounted for in this estimation, meaning that costs associated with more sensitive ecosystems may have been overlooked.

3.5 Animal welfare

The Animal Welfare Index in this study is based on the Perception Adjusted Animal Lives Affected (PAALA) metric, as described by Röös et al [38]. PAALA quantifies perceived animal suffering per kilogram of commodity, making it particularly suitable for dietary assessments.

The PAALA index is constructed from three components:

- Number of animals affected per kilogram of food produced
- Perceptual ability of the animal species to experience suffering

- Degree of suffering, including factors such as exposure to disease and stress during the production system, as well as the impact of slaughter

The formula for calculating PAALA is as follows:

$$\text{PAALA} = \text{number affected} \times \text{ability to perceive suffering} \times$$
$$\text{suffering (including slaughter)} \times 100 \quad (3.9)$$

Although similar to the more conventional metric Animal Life Years Suffered (ALYS), PAALA differs by expressing perceived suffering per kilogram of food, thus offering a more consumption-oriented perspective. The indices are shown in Figure 3.3.

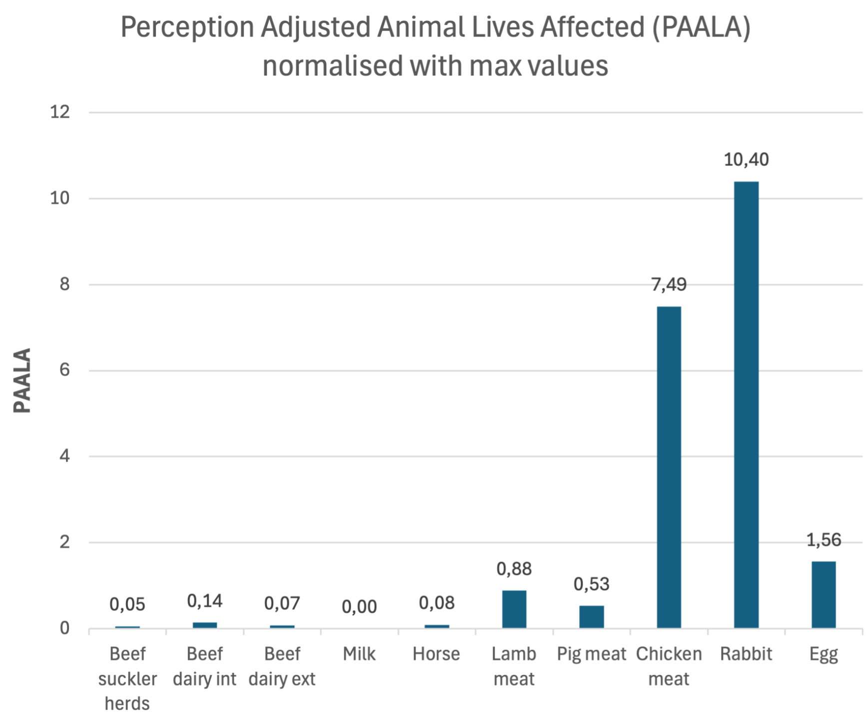

Figure 3.3: PAALA index for 1 kilogram of livestock products produced in Sweden

In terms of valuation, the ALYS metric has been compared to the human DALYs framework, which allows for a monetisation of welfare impacts for animals based on the value of life quality. Including the value of animals in cost–benefit analyses is significant, as it acknowledges the intrinsic worth of non-human lives. Based on a WTP estimation for data regarding poultry, the value of animal welfare per affected unit has been approximated at USD 0.10 to USD 0.37 per kilogram [65]. However, relying solely on human WTP of other species may severely underestimate the true intrinsic value since WTP reflects human preferences, awareness and biases. This implies that any attempt to construct social welfare functions for valuing animal welfare, necessarily reflects underlying ethical and philosophical assumptions, highlighting the normative choices embedded in welfare aggregation across species [66]. Kuruc and McFadden [67] present an example of a generalised totalist utilitarian welfare function, following traditional economist frameworks. In their analysis, the value of animal welfare impact is calculated to USD 122,789.19, which is about SEK 1,2 million for annual meat consumption of an average non-vegetarian diet for an American. This is not representative for Sweden since dietary patterns differentiate between countries, but it can act as a perception of magnitude. The model assigns animals a fixed negative utility value across species, based on the ethical assumption that lives in industrial farming conditions are worse than non-existence. This is operationalised in the model by setting animal welfare at a human-equivalent utility level of USD 1.00 per day, below the international human poverty line of USD 1.90 per day. These utility losses are then aggregated across the number of animal life-years required to sustain typical meat consumption, yielding a total estimate of the external welfare cost.

An alternative monetisation method involves assessing the additional costs required to improve animal welfare, such as increased living space, better nutrition, and improved health, thereby reduce suffering. However, such cost estimates have not yet been quantified in detail and are therefore not monetised in this thesis. A formal model addressing this issue has been developed, providing a theoretical basis for understanding the economic impact of animal welfare improvements at slaughter [68]. While valuing animal welfare strictly through damage cost approaches offers practical advantages for integration into economic analyses, it also raises important ethical concerns. Reducing the intrinsic value of animal life to market prices risks marginalising the moral worth of sentient beings.

These considerations led to the conclusion that the available monetisation methodologies are not sufficiently reliable for this analysis. Consequently, this impact area was excluded from the results presented in this thesis.

3.6 Health impacts

This section presents the methods used to quantify health-related impacts associated with the Swedish food consumption patterns. It includes five distinct health indicators: dietary risks, pesticide exposure, air pollution, heavy metal exposure, and antibiotic use. Each indicator required a tailored methodological approach, depending on the type and availability of the data.

An overview of the included indicators and their methodological basis is provided in Table 3.7, with further detail in the subsections that follow.

Table 3.7: Methods for economic valuation of health-related externalities by indicators for Swedish food consumption

Indicator	Calculation
Dietary risks	DALY estimates from GBD 2021 [69], based on associations between dietary intake and disease outcomes
Pesticide use	Average cost per kg a.i. derived from Zandonella et al. [70], applied to pesticide use in food consumed in Sweden
Air pollution	DALY estimates from GBD 2021 [69] for PM and ambient ozone, and characterisation factors (μDALY/kg emitted) for NH_3 from Humbert [71]
Antibiotic use	DALYs lost due to infections with antibiotic-resistant bacteria from the European Centre for Disease Prevention and Control [72]
Heavy metal exposure	DALY estimates per contaminant (lead, inorganic arsenic, cadmium, and methylmercury) from Thomsen et al. [73] and monitoring data from Petersen et. al. [74]

3.6.1 Dietary risks

To estimate the disease burden linked to diet, data from GBD 2021 [69] was used. For dietary risks, GBD provides point estimates of DALYs attributable to each dietary risk in Sweden for the year 2021. These estimates represent the best central value based on 1,000 simulations and reflect Swedish dietary patterns, disease rates, and demographics [42].

To allocate these burdens to specific foods for the calculation of annual and per-kilogram externality costs, GBD risk factors were matched to food categories in the modified SAFAD dataset by Larsson et al. [30]. As shown in Table 3.8, not all GBD dietary risks could be clearly mapped to the food categories. Only dietary risks that could be directly linked to specific food groups were included in the category-level calculation, while others were excluded to avoid double-counting or ambiguity.

Table 3.8: Mapping of GBD dietary risk factors to food categories

GBD Dietary risk factor	Mapped to food category	Reason if not mapped
Diet low in fruits	Fruits	-
Diet low in vegetables	Vegetables	-
Diet low in legumes	Legumes	-
Diet low in whole grains	Whole grain categories	-
Diet low in nuts & seeds	Nuts & seeds	-
Diet low in milk	Milk	-
Diet low in fiber	Not mapped	Difficult to isolate source, overlaps with multiple food categories
Diet low in calcium	Not mapped	Already indirectly captured via "low milk"; risk of double-counting
Diet low in seafood omega-3	Fish & shellfish	-
Diet low in polyunsaturated fatty acids	Not mapped	Spread across multiple oils/fats, unclear mapping
Diet high in red meat	Two categories: Beef & lamb, Pork	-
Diet high in processed meat	Processed meat	-
Diet high in sugar-sweetened beverages	Sugar-sweetened softdrinks	-
Diet high in sodium	Not mapped	Affects many foods; cannot allocate to single category
Diet high in trans fatty acids	Not mapped	Trans fats are ingredient-level; difficult to assign to specific category

The DALY values obtained from GBD was used to estimate the annual health-related costs for each food category. For red meat such as *beef & lamb* and *pork*, consumption data from the modified SAFAD dataset was used to proportionally allocate the burden between the two categories. This was possible because the disease burden results from overconsumption, allowing the cost to be distributed according to the relative intake of each type of meat. In the case of whole grain products, the burden originates from insufficient intake. Since the health benefit is associated with increased consumption of the entire food group, it is not meaningful to separate the effect of separate whole grain categories such as *bread* and *pasta*. The deficiency reflects an overall lack of whole grain consumption, regardless of which specific product is underconsumed. Therefore, the burden was not divided across subcategories but instead attributed to whole grain products as an aggregate.

To estimate DALYs per kilogram of food, dietary intake data from GBD (g/day by age group 25+) [75] was combined with population data from SCB [76] to produce a population-weighted average daily intake. This was then scaled up to the entire Swedish population to calculate annual national intake in kilograms. Although GBD provides data only for individuals aged 25 and above, the burden was distributed across the entire population. This decision was based on evidence that diet-related diseases develop over time [77] and that early-life dietary patterns are closely linked to later health outcomes [78].

The TMREL values defined by the GBD were used as benchmarks for optimal intake levels, as presented in Table 3.3. For foods with excessive intake, the lowest value in the TMREL range was used to calculate the deviation. This allocated the DALYs across the largest possible intake surplus, resulting in a conservative per-kilogram cost estimate:

$$\text{DALY/kg (excess)} = \frac{\text{Total DALYs}}{\text{Actual intake} - \text{Lower TMREL bound}} \tag{3.10}$$

For foods with insufficient intake, the highest value in the TMREL range was used. In the case of *diet low in milk*, which had separate TMRELs for males and females, the average of the two upper bounds was applied. This allocated the DALYs across the largest possible deficiency, again resulting in a conservative cost:

$$\text{DALY/kg (deficiency)} = \frac{\text{Total DALYs}}{\text{Upper TMREL bound} - \text{Actual intake}} \tag{3.11}$$

Each DALY-per-kilogram value was then assigned to the corresponding food group in SAFAD. A positive value indicates a health cost due to overconsumption, while a negative value represents a health benefit from addressing underconsumption.

This method assumes a linear relationship between intake deviation and health impact, meaning that each additional kilogram above or below the optimal intake contributes proportionally to the total DALY burden. The resulting DALY/kg value can be interpreted as the marginal health burden of consuming one extra, or one less, kilogram of a given food. This assumption is made to be able to produce a single value, even though GBD applies non-linear dose response functions [42]. It is therefore of importance to note that the method does not account for the diminishing or accelerating risk effects that may exist at different intake levels.

Moreover, while GBD estimates DALYs using the full intake distribution, the per-kilogram analysis here was limited to risks where the national average intake lies outside the TMREL range. Consequently, the risk factor *diet low in seafood omega-3* which could be mapped to *fish & shellfish*, was excluded from the per-kilogram analysis since the mean Swedish intake already falls within the optimal range. Including it could overstate the societal cost of average consumption, as the remaining burden likely arises from specific subpopulations.

Finally, to estimate the total national burden from diet-related diseases, the DALY value reported by GBD for all dietary risks combined was used, rather than limiting the sum to only those that could be mapped to SAFAD. This avoids double-counting and ensures inclusion of all the dietary risks in GBD.

3.6.2 Pesticide use: Human toxicity

To estimate the health-related costs of pesticide exposure, this study uses a Swiss analysis by Zandonella et al. [70], which provides both consumption data and monetised health impact estimates. A comparison with Swedish data from Cederberg et al. [79] indicates that the proportional use of herbicides, fungicides, and insecticides is relatively similar in the two countries. Since these pesticide types differ in their potential health impacts [60], this similarity supports the use of the Swiss estimates as a reasonable proxy for Swedish conditions.

A consumption-based approach was used, meaning the estimates reflect the health impact of pesticides used in the production of food consumed in Sweden, regardless of where the pesticides were applied geographically. It should be noted that applying the Swiss health cost estimate to pesticide use associated with Swedish food consumption assumes that the health impacts per kilogram of pesticide are similar across countries. This introduces uncertainty. Since the consumption-based approach includes both domestically produced and imported food, the actual health costs might differ depending on where the pesticide exposure occurs. In some cases, health-related costs in production countries could be lower due to weaker healthcare systems, regulatory environments, or lower valuation of health outcomes, whereas in other cases they could be higher due to less protection for agricultural workers or greater exposure risks. Both Sweden and Switzerland have high levels of human development [80] and per capita health spending [81], supporting the assumption that the health impact per unit of pesticide use could be similar in both contexts. However, Switzerland has a slightly higher Human Development Index (HDI) [80] and marginally higher health expenditures per capita [81], which could lead to an overestimation of the costs. Furthermore, the analysis assumes a linear relationship between pesticide use and associated health costs, although this is likely a simplification that does not fully reflect the complexity of real-world exposure-response dynamics.

Zandonella et al. [70] provides several cost estimates derived from different methodological approaches. To account for this variation, the average of these estimates was used to calculate a representative cost per kilogram of a.i.

$$\text{Cost/kg a.i.} = \frac{\text{Total Swiss health cost (CHF)}}{\text{Total Swiss pesticide use (kg a.i.)}} \tag{3.12}$$

This unit cost was then used in a general formula to estimate health related costs:

$$\text{Cost} = \text{Cost/kg a.i.} \times \text{Pesticide use (kg a.i.)} \tag{3.13}$$

This calculation was performed both for total pesticide use linked to Swedish food consumption, and by food group. The pesticide consumption data was obtained from the SAFAD dataset.

3.6.3 Air pollution

Air pollution is complex and often difficult to trace or quantify. A major limitation in assessing its health impact from agriculture is the lack of detailed data on emissions, dispersion, and exposure pathways. In particular, the formation of particulate matter (PM) further complicates matters, as its dynamics make it difficult to establish clear links between emissions and health outcomes. Air pollution, and especially $PM_{2.5}$, is linked to respiratory and cardiovascular diseases, as well as premature death [82]. Agricultural emissions, such as ammonia, contribute significantly to the formation of secondary PM, making them a notable public health concern [83].

The calculations used to estimate air pollution impacts from agriculture is simplified and generalised, focusing on broader assumptions rather than specific geographical data. These models may overlook complexities in atmospheric chemistry, pollutant dispersion, and long-range transport of secondary pollutants as stated above.

The following method was used to estimate the total DALYs associated with air pollution from agriculture in Sweden:

Firstly, data from GBD regarding the total health burden in Sweden due to air pollution was collected [69]. The data included estimates of DALYs attributed to PM, ambient ozone pollution and nitrogen dioxide pollution.

The GBD dataset provided the total DALYs from air pollution in Sweden, allowing for an overview of the health impacts at a national level. The share of each specific pollutant related to agriculture was identified by analysing available sectoral emission data from the EEA [84]. The total DALYs from agriculture-related air pollution were estimated by multiplying the proportion of each pollutant attributable to agriculture by the total DALYs associated with that pollutant, as shown in the equation below:

$$\text{Total DALYs}_{\text{agriculture}} = \sum_{i=1}^{n} (\text{Share}_i \times \text{Pollutant}_i) \tag{3.14}$$

However, for ambient ozone and nitrogen dioxide pollution, certain considerations and limitations were applied. For ambient ozone formation, a simplified 1:1 ratio between non-methane volatile organic compounds (NMVOCs) and NO_x was applied, deviating from the more typical 70:30 ratio commonly used for urban environments [85]. Since agriculture is predominantly rural, where ozone formation tends to be NO_x-limited rather than NMVOC-limited, this simplification was made to better reflect the atmospheric chemistry in those areas. The share of pollutants was then calculated using a newly weighted average, which was later used in the health impact calculations.

Nitrogen dioxide was not included in the report primarily due to a lack of specific data related to its agricultural emissions. Additionally, its health impacts are more relevant for urban sectors such as traffic and industry, rather than for agriculture. Furthermore, since nitrogen dioxide is included in NO_x and acts as a precursor to ozone rather than a direct pollutant, its inclusion would not significantly affect the analysis.

Since ammonia emissions were not included as a separate category in the GBD data, the health impacts from ammonia emissions was calculated separately. This was done by taking the total national emissions and multiplying them by a factor representing the health damage per kilogram emitted in rural areas, expressed as μDALY/kg emitted pollutant [71]. This approach allowed for an estimation of the impact of ammonia using emission-specific health impact factors appropriate for rural settings.

Due to limited data availability regarding emissions per food item, the calculations for the individual food categories are mostly based on ammonia emissions, and to a lesser extent NMVOCs. Data for emissions of the other pollutants per kilogram of food was not found, which limited the analysis.

The annual cost per capita for each category was calculated using the following equation:

$$\text{Cost per capita/year} =$$

$$\sum \left(\frac{\text{kg Pollutant}}{\text{kg food}} \times \text{kg food/person/year} \times \frac{\text{DALY}}{\text{kg Pollutant}} \times \text{SEK/DALY} \right) \quad (3.15)$$

These results were also multiplied by the total population for the, giving the total annual cost from these emissions.

Moreover, since Sweden imports approximately 60% more agricultural products and food than it exports [86], it is assumed that the emissions per unit of food are similar for imported and domestically produced products. To reflect that Sweden consumes more food than what is produced domestically, the territorial external costs are adjusted using a factor of 1.6. This factor represents the ratio between total food consumption in Sweden, using both domestic production and net imports, but also the domestic production alone. This adjustment for import and exports is only done for the total cost calculation and not for the food categories, since those are consumption-based and already adjusted.

3.6.4 Antibiotic use

The health burden of antimicrobial resistance (AMR) due to antibiotic use was assessed using the antibiotic index from the SAFAD database. National data on antibiotic sales for veterinary purposes in different EU member states is collected annually through the ESVAC project [87]. However, data on the actual use of these products, as well as species-specific data, is still lacking, as member states are currently not required to report antibiotic use data by animal species for food-producing animals. Because of this, the calculations of the antibiotic index are based on the assumption that the amount of sold antibiotics mirrors the amount of antibiotics used [88].

The SAFAD antibiotic use index was calculated by comparing the total sales of antibiotics in veterinary medicinal products across different EU countries with the expected sales based on the proportion of livestock species and their biomass [88]. For each country, the expected sales were derived by multiplying the proportion of each species biomass in that country by the standardised antibiotic usage values (mg/kg biomass) for each species. These standardised values reflect the average amount of antibiotics used for each species, based on available data from six European countries. The resulting ratio of total sales to expected sales was then multiplied by the standardised value for each species, as seen in Equation 3.16, producing the index. This approach enables for estimating antibiotic use based on national differences in species distribution, even when direct usage data is unavailable. In this report, the antibiotic index is used as an estimate of the amount of antibiotics used (mg) to produce one kilogram of each food category.

$$\frac{\text{Total sales}}{\text{Expected sales}} \times \text{Standardised usage value} = \text{Antibiotic index} \tag{3.16}$$

The yearly health cost of infections caused by antibiotic-resistant bacteria, measured in DALYs, for all EU member states was collected from the European Centre for Disease Prevention and Control [72]. An estimate of the health burden directly related to food consumption was then made based on the assumption that 22% of all AMR is attributable to the food system [89]. This number was then divided by the amount of antibiotics sold for veterinary purposes in the EU to give an estimate of the DALYs lost to AMR per milligram of antibiotics, using data on antibiotic sales collected from the European Medicines Agency [90].

The calculations are based on the assumption that all use of antibiotics contribute equally to AMR, which is a simplification. In reality, the spread of AMR depends on numerous different factors, such as the type of antibiotics and the way it is used, which exceeds the scope of this report.

3.6.5 Heavy metal exposure

To estimate the disease burden from foodborne heavy metals, two different methods were used to calculate the total disease burden and the burden per food category. These methods relied on studies conducted in Denmark. Due to the lack of Swedish data and given the geographical and cultural similarities between the two countries, it was assumed that the Danish data was a reasonable proxy for Swedish conditions.

For the total disease burden, a study by Thomsen et al. [73] was used. It estimated DALYs attributable to four chemical contaminants commonly found in food: inorganic arsenic (i-As), lead (Pb), methylmercury (MeHg), and cadmium (Cd). The associated health outcomes include intellectual disability, chronic kidney disease, and various forms of cancer, such as lung, bladder, and skin cancer. The burden was calculated using monitoring data and expressed as DALYs per 100,000 individuals.

In the case of Pb, the study provided two estimates based on uncertainty in the conversion factor between dietary intake and blood lead levels [73]. The lower bound resulted in 6.6 DALYs/100,000, while the upper bound was 22 DALYs/100,000. These differences arise due to variation in how efficiently dietary lead is assumed to transfer to blood lead levels. To derive a single estimate, the mean value between the lower and upper bound was used in this analysis. These DALY rates were scaled up to the Swedish population to estimate the total disease burden from heavy metal exposure via food.

For disease burden by food category, both the study by Thomsen et al. [73] and a report by Petersen et al. [74] were used. The latter monitored food contaminants between 2004-2011, presenting these in micrograms of contaminant per kilogram of food (μg/kg) for the same four heavy metals.

The DALY per microgram of contaminant (DALY/μg) was calculated using the following formula:

$$\frac{\text{Total DALY/year}}{\text{Average exposure/day} \times \text{Average weight} \times \text{Population} \times 365} = \text{DALY}/\mu\text{g} \quad (3.17)$$

Both the total DALYs per year and the average daily exposure for the four chemicals were sourced directly from the report by Thomsen et al. [73]. The average exposure was defined as the amount of contaminant per kilogram bodyweight per day. However, only Cd and i-As exposure was calculated for the whole population, Pb exposure data was limited to 5-year olds and MeHg exposure was calculated only for women in fertile age (15-49 years old) [73]. Accordingly, the *Population* and *Average weight* parameters varied for each chemical. For MeHg, Cd and i-As an average weight of 70 kilogram was assumed, while for Pb, a weight of 21.5 kilogram was assumed. The subpopulation sizes for the year 2019 were obtained from Statistics Denmark [91]. These values are shown in Table 3.9.

Table 3.9: Subpopulations affected by specific chemical exposures in the study by Thomsen et al. [73], including population size and assumed average body weight used in the calculations

Chemical	Subpopulation	Size	Assumed weight
Pb	5-year olds	58,352	21.5 kg
MeHg	Women 15-49	1,267,296	70 kg
Cd	Entire population	5,806,081	70 kg
i-As	Entire population	5,806,081	70 kg

These DALY values were then compared to the numbers in the study by Petersen et al. [74]. By multiplying the amount of DALYs associated with each chemical with the amount of chemical found in different food groups, a number of DALYs per kilogram was generated for each available food category.

3.7 Monetisation

In order to compare and aggregate the various external costs, it was necessary to assign monetary values to each externalities. To ensure consistency across monetary values, all foreign currencies were converted to Swedish kronor (SEK). Historical average exchange rates from the year in which the original amount was reported were used for the conversion. If the specific year was not stated, the average rate from the year the report was published was applied instead. All exchange rates were obtained from the official database of Sweden's central bank (Sveriges Riksbank) [92].

To account for inflation, all amounts were subsequently adjusted to 2024 values using the price conversion function of SCB (Prisomräknaren) [93]. This tool adjusts for inflation based on the Consumer Price Index (CPI), allowing for a consistent comparison of monetary values across different years.

Table 3.10 summarises the final monetised values applied in this study for each environmental and health-related externality by indicator. All values presented in the table reflect the amount after currency conversion and inflation adjustment.

Table 3.10: Monetisation of environmental and health externalities by their indicator, at 2024 price level

Externality indicator	Unit	SEK/unit	Source
Carbon footprint	kg CO_2e	2.16	EPA, 2023 [43]
New N input	kg N	44.28	Swedish Environmental Protection Agency, 2018 [44]
Pesticide use	kg a.i.	19.5	Leach and Mumford, 2008 [45]
Ammonia emissions	kg NH_3	6.77	Swedish Environmental Protection Agency, 2018 [44]
Cropland use	MSA · ha · year	12,761	True Price, 2023 [46]
Dietary risks	DALYs	1,250,800	True Price, 2023 [46]
Pesticide use	kg a.i.	225	Zandonella et al., 2014 [70]
Air pollution	DALYs	1,250,800	True Price, 2023 [46]
Antibiotic use	DALYs	1,250,800	True Price, 2023 [46]
Heavy metal exposure	DALYs	1,250,800	True Price, 2023 [46]

3.8 National quantification

National quantification of the true cost of food consumption in this thesis is achieved by summing current national food expenditure with the aggregated monetised external costs.

$$\text{True Cost} = \text{National Food Expenditure} + \sum_{i=1}^{n} \text{Externality}_i \tag{3.18}$$

When necessary, the Swedish population from 2021 (10,452,326 individuals) [76] was used to scale externalities to the national level. This aligns with GBD, while the SAFAD dataset reflects conditions for 2024. Although this means that data sources refer to slightly different years, the difference in population size between 2021 and 2024 is marginal [76]. Given the approximate nature of the estimates used throughout this report, the impact of this simplification on the final results is negligible.

The national food expenditure was estimated by multiplying the average annual food spending per household by the total number of households in Sweden, based on data from SCB [94]. While this provides insight into current household-level spending, these expenditures include VAT. Since VAT is a transfer payment between consumers and the public sector, and therefore not a net cost to society, it should not be considered a societal cost in an economic sense. Therefore, the VAT portion of the national expenditure was excluded. In Sweden, the VAT on food is 12% [95], meaning that the reported national expenditure should be adjusted as in the equation below.

$$\text{Net expenditure (excl. VAT)} = \frac{\text{Total expenditure (incl. VAT)}}{1 + \text{VAT rate}} \tag{3.19}$$

To determine the average annual per capita food expenditure, the VAT-adjusted household food expenditure was divided by the average number of persons per household, as shown in the equation below:

$$\text{Per Capita Food Expenditure} = \frac{E}{P} \tag{3.20}$$

where E = average annual food expenditure per household (excl. VAT)
P = average number of persons per household

3.9 Food group quantification

To calculate the true price for individual food categories, average market prices were combined with externality prices. These average market prices were primarily derived from data provided by Larsson et al. [30]. For some categories, however, no specific prices could be extracted. In those cases, current market prices were obtained from three major Swedish retail chains: ICA [96], Willys [97], and Coop [98].

3.10 Sensitivity Analysis

To determine how variations in the valuations of both environmental and health externalities affect the results, a sensitivity analysis was conducted. Externalities are often difficult to monetise due to their complexity, resulting in different valuations. This can significantly impact both the final outcomes and the reliability of the estimated total cost. In this report, the sensitivity analysis focuses on DALYs, since they are used for most health-related externalities, as well as on the environmental externalities with the highest total values, as these factors have the strongest influence on the overall result.

The different valuations for DALYs are presented in Table 3.11. The valuation used in the results is based on estimates from True Price [46], while the alternative values used in the sensitivity analysis originate from the Swedish Environmental Protection Agency [44]. These were initially expressed as QALYs and later converted to DALYs, assuming that the monetary valuation of a QALY is relatively similar to that of a DALY. This approach was adopted to allow the use of Swedish data for monetising DALYs, despite the considerable uncertainty associated with assuming that the valuations of DALYs and QALYs are comparable.

Table 3.11: Valuation of a DALY from various sources, converted to SEK (2024 price level)

Source	Price per DALY (SEK)
True Price [46]	1,250,800
Swedish Environmental Protection Agency (Lower) [44]	635,910
Swedish Environmental Protection Agency (Upper) [44]	7,021,515

The three largest environmental externalities identified in the result are carbon footprint, cropland use, and new N input. Due to the limited availability of valuation methods for cropland use, the sensitivity analysis was only conducted for the other two categories.

Table 3.12 presents the total cost of carbon emissions under three different discount rates, in accordance with EPA estimates for the SCC [43]. The results illustrate how variations in the discount rate, which reflect differing assumptions about future damages, as described in Section 3.4.1, can lead to substantial differences in the estimated societal costs.

Table 3.12: Estimated societal costs of carbon emissions for yearly Swedish food consumption at varying discount rates and the respective share of Total National Expenditure (TNE)

Discount rate	SEK/kg CO_2	Total cost (billion SEK)	% of TNE
1.5%	3.87	60.7	5.3%
2.0%	2.16	33.9	3.0%
2.5%	1.36	21.4	1.9%

Table 3.13 presents the values for nitrogen used in the sensitivity analysis. The monetisation factor for nitrogen emissions into marine waters from True Price [46] was compared to the valuation (see Equation 3.5) derived from the Swedish Environmental Protection Agency [44].

Table 3.13: Valuation of nitrogen emissions from various sources, converted to SEK (2024 price level)

Source	Price per kg N emissions (SEK)
True Price [46]	181
Swedish Environmental Protection Agency [44]	44.3

4

Results

The following section presents the results of this study, detailing both the total true costs associated with Swedish food consumption and the specific costs attributed to different food categories. In addition, the uncertainty ranges from the sensitivity analysis are presented.

4.1 Total true cost of Swedish food consumption

Figure 4.1 illustrates the total true cost of the Swedish food system, including both the national expenditure and the estimated external costs. The national expenditure, as defined in Equation 3.19, represents the total amount Swedish households spend on food annually. The national food expenditure was estimated at approximately SEK 336 billion (see Section 3.8). This corresponds to an average annual per capita food expenditure of around SEK 36,000, which equates to a monthly cost of about SEK 3,000 per person. In comparison, the external costs were estimated at approximately SEK 302 billion, of which SEK 57 billion corresponds to environmental and SEK 245 billion corresponds to health.

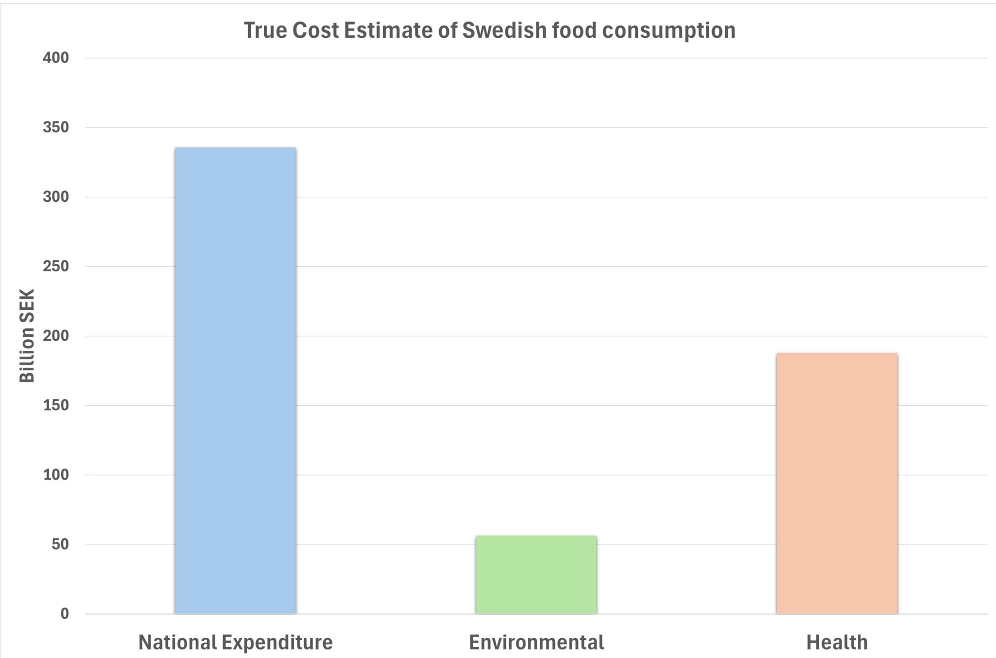

Figure 4.1: True cost of Swedish food consumption, including national expenditure along with external costs from environmental and health impacts

While Figure 4.1 provides an overview of the total costs, Table 4.1 details the specific external costs associated with various environmental and health-related indicators. Among these, dietary risks stand out as the most significant external cost, amounting to approximately SEK 235 billion. This estimate is by far the largest compared to the other external costs. The costs related to the environment are significantly lower compared to health, representing approximately 19% of the total sum of external costs.

Table 4.1: National market expenditure and estimated externality costs by environmental and health-related indicators

Externality Indicator	**Estimated Cost (Million SEK)**
Carbon footprint	33,942
New N input	3,915
Pesticide use	75
Ammonia emissions	452
Cropland use	18,133
Total Environmental Cost	**56,517**
Dietary risks	234,695
Pesticide use	867
Air pollution	4,158
Antibiotic use	2,402
Heavy metal exposure	3,177
Total Health Cost	**245,299**
Sum of External Costs	301,816
National Food Expenditure	335,805
National True Cost	**637,621**

4.1.1 Environmental costs

As shown in Figure 4.2, carbon footprint dominates the environmental impact, accounting for approximately 60% of the total cost. Among the remaining externalities, cropland use has the highest contribution with 32%, followed by new N input at 7%.

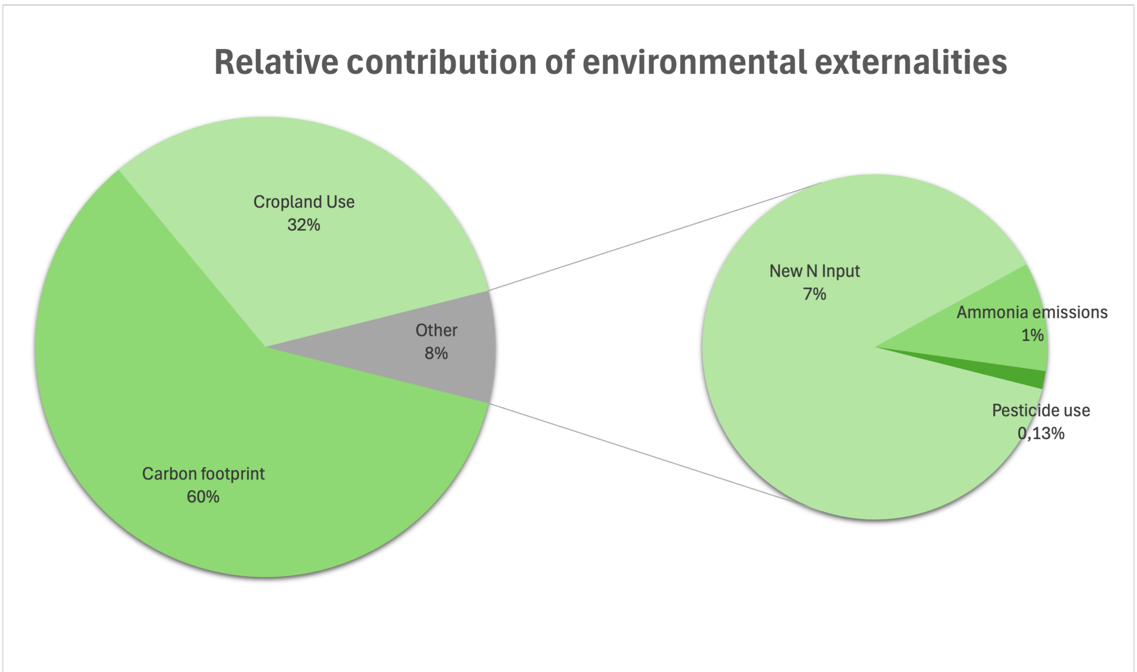

Figure 4.2: Share of different environmental externalities (by indicator) contributing to total environmental impact

4.1.2 Health costs

Figure 4.3 illustrates the proportion of each externality relative to the total health-related costs. As previously noted, dietary risks are by far the largest contributor of the costs, accounting for approximately 96% of the total health costs. The other remaining 4% are more evenly distributed among air pollution, heavy metals and antibiotics, which have relatively similar external costs. While their impacts still differ by several million SEK, they are significantly smaller compared to the dominant cost of dietary risks.

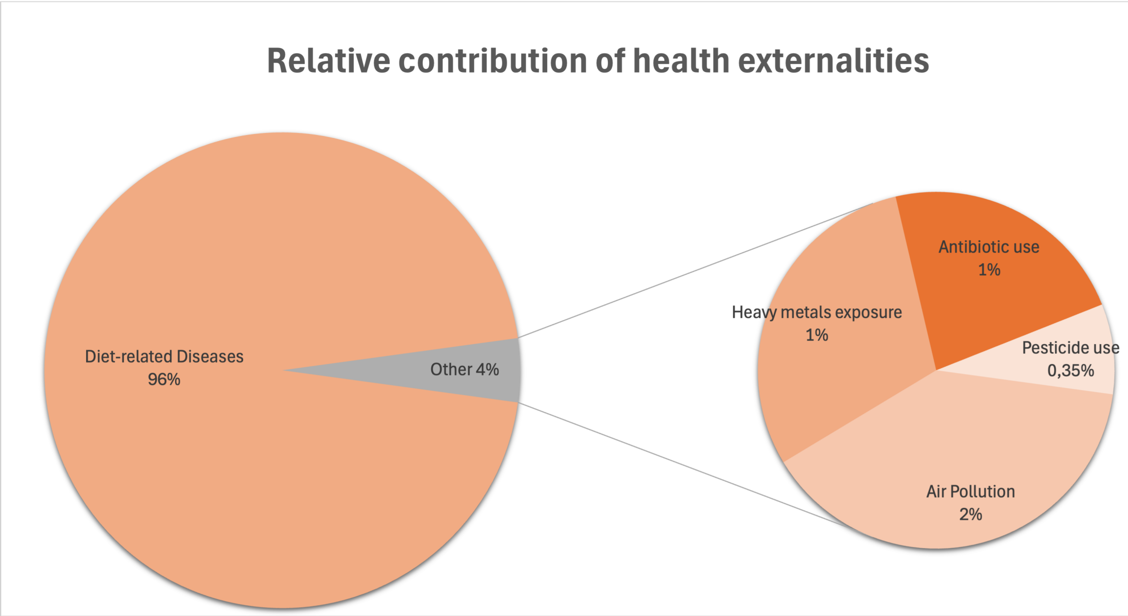

Figure 4.3: Share of different health externalities (by indicator) contributing to total health impact

4.2 True cost per food group

Figure 4.4 illustrates how different types of externalities affect the estimated cost per kilogram for selected food categories. The bars show the market price along with the separate contributions from environmental and health-related externalities.

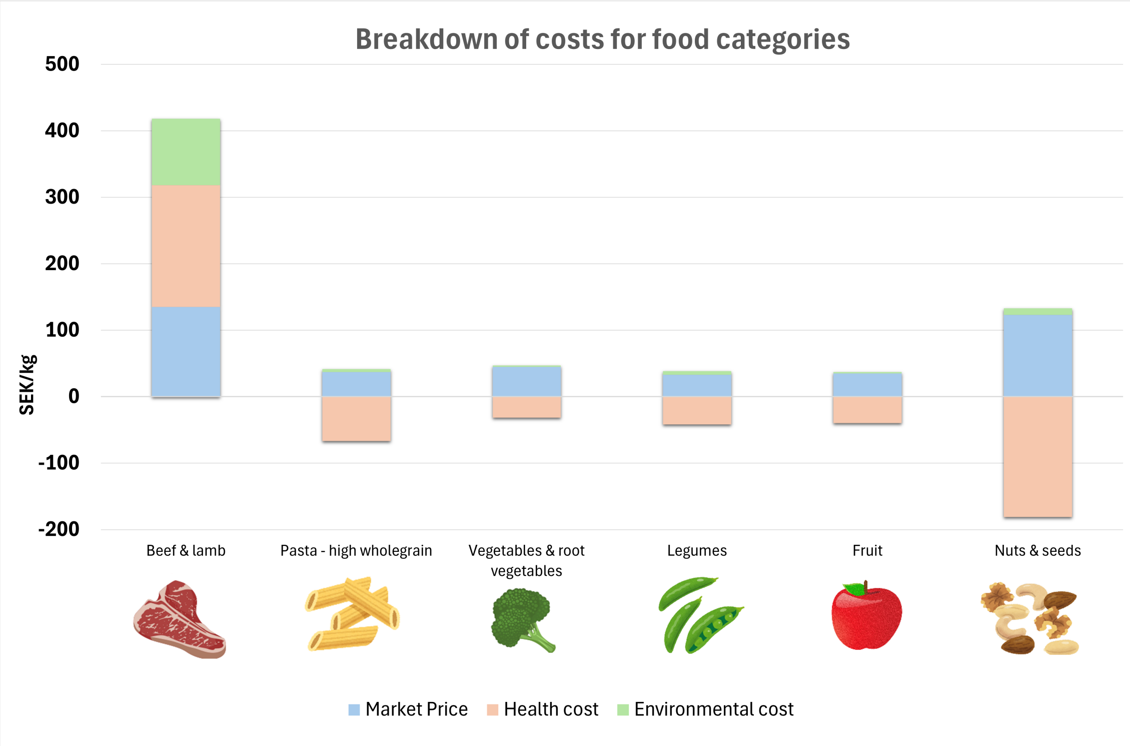

Figure 4.4: Market price, environmental and health-related external costs per kilogram for selected food categories

The net true costs of the selected food categories can be seen in Table 4.2.

Table 4.2: Net true costs for selected food items (rounded)

Food category	Market price (SEK/kg)	Env. cost (SEK/kg)	Health cost (SEK/kg)	True cost (SEK/kg)
Beef & Lamb	135	100	183	418
100% wholegrain pasta	37	5	-67	-25
Vegetables & Root vegetables	45	2	-32	16
Legumes	33	6	-42	-3
Fruits	35	2	-40	-2
Nuts & Seeds	123	10	-181	-48

The result shows that animal-based foods carry significant external costs that are not accounted for in the market price. According to the estimation, the price of *beef & lamb* should increase by approximately 175% to reflect the true costs. Many plant-based foods, on the other hand, appear to be more sustainable, which results in their true costs being lower than their current market prices. The negative cost is obtained from the health cost column representing net health benefits. The results show that price adjustments are highly dependent on the cost of diet-related diseases.

While the per-kilogram cost reflects the marginal external cost of consuming an additional kilogram of a given food, the total annual costs presented in Table 4.3 are based on current consumption levels and illustrate the overall societal burden of each food category. Based on these annual costs, the food categories with the greatest environmental costs were *beef & lamb*, *processed meat*, *cheese* and *coffee, tea & cocoa.* In contrast, the food groups with the highest total health-related costs were *wholegrain products*, *processed meats*, red meats (including *beef & lamb* and *pork*), and *fruit.* The top contributors to overall externality costs are shown in Table 4.3, while a full overview of costs associated with all aggregated food categories is provided in Appendix A.

Table 4.3: Estimated annual national costs of food categories

Food category	**Estimated annual cost (Million SEK)**
Wholegrain products*	67,420
Processed meat	58,734
Beef & lamb	34,058
Fruit	31,890
Vegetables & root vegetables	28,415
Pork	23,063

* *Wholegrain products* includes aggregated costs from several categories with a high wholegrain content: *bread*, *pasta*, *porridge oats*, and *muesli & cereals*

4.3 Sensitivity analysis

Figure 4.5 presents the total societal costs per year for five outcomes, divided into environmental and health externalities. The outcomes are based on different combinations of the following valuation scenarios:

- The Mid scenario reflects the baseline values used throughout this report and represents the central estimate.
- The High Health and Low Health scenarios apply the upper and lower DALY valuations from Table 3.11, respectively.
- The High Env. scenario applies the upper values for carbon and nitrogen externalities, based on the ranges shown in Tables 3.12 and 3.13.

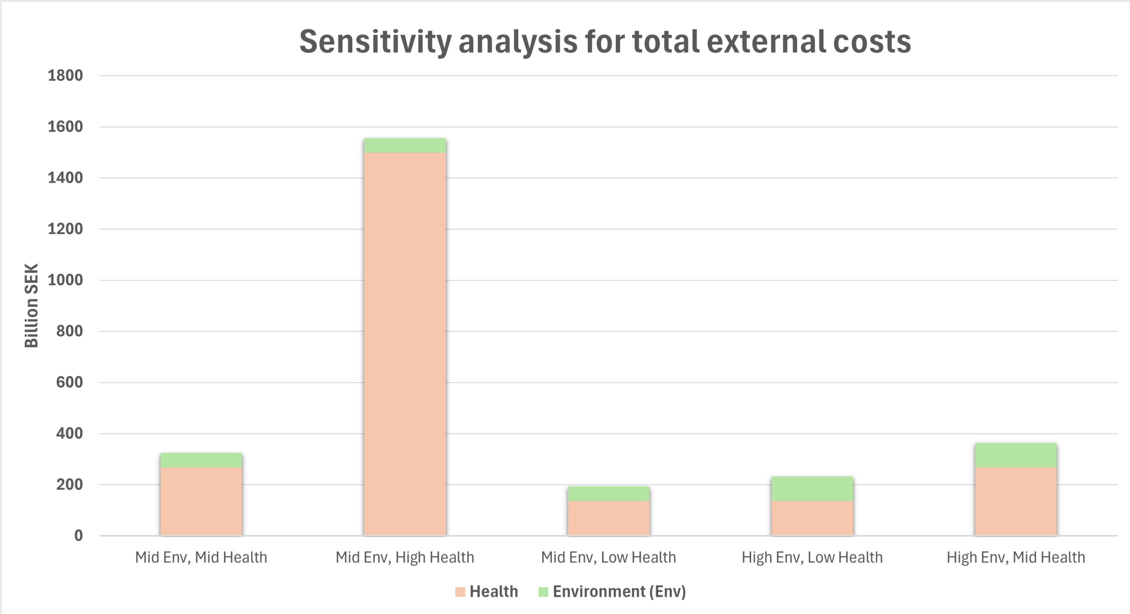

Figure 4.5: Sensitivity analysis of the total societal cost of food under alternative health and environmental valuation scenarios

The sensitivity analysis of the total externality costs shows noticeable variation in the results depending on the input values used. The valuation applied to DALYs has the largest impact on the total estimated costs. This is expected, as health-related costs constitute a larger proportion of the externalities in the baseline scenario, and the valuation range for DALYs is broader than that for carbon footprint. When a higher DALY value is applied, the health-related cost estimates increase substantially, leading to a total cost estimate nearly six times higher than the baseline. Conversely, applying the lower DALY valuation reduces the health-related estimate to roughly half of the baseline level. For environmental impacts, using higher input values results in an estimated total that is approximately twice the baseline. While this represents a significant increase, the variation is less sensitive compared to the uncertainty range for health-related costs.

The sensitivity analysis for the selected food categories in Figure 4.6 is performed using alternative values for DALYs, as health costs represent the largest share of total externalities. For simplicity, environmental variation is not included. As a result, certain external costs, such as the carbon footprint of *beef & lamb*, are not reflected in this analysis.

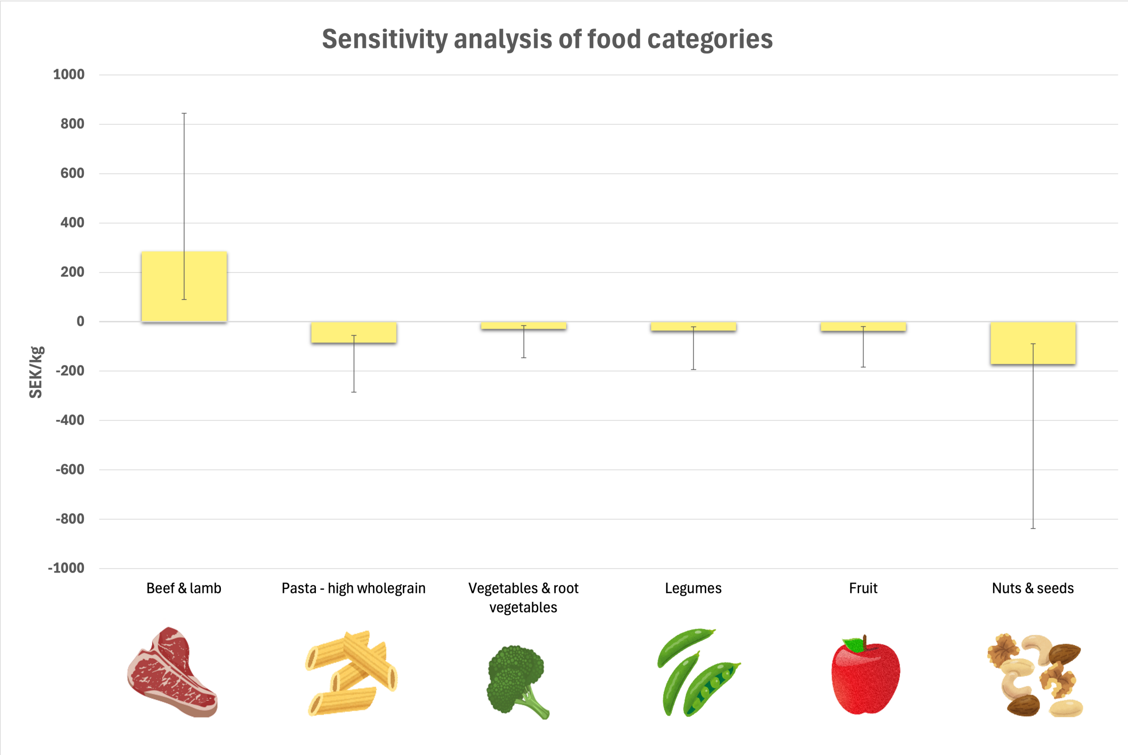

Figure 4.6: Sensitivity analysis of external costs by food category. Bars represent baseline estimates of external costs, while whiskers illustrate the range resulting from applying low and high DALY valuations

Figure 4.6 shows that the uncertainty ranges for the selected food categories are relatively large. The upper estimates, based on higher DALY monetisation values, lead to substantial increases in external cost estimates. For example, the external cost of *beef & lamb* reaches approximately SEK 850 per kilogram, which is SEK 565 higher than the baseline value. In contrast, the lower estimate deviates less from the baseline, with a difference of around SEK 195. This reflects an asymmetry in the uncertainty range, where the upper bound contributes more significantly to the total variation.

5
Discussion

In this chapter, the results and uncertainties of this thesis will be discussed, as well as their implications for future research and policy interventions. The following questions will be addressed:

- *How does the result of the different food categories compare to each other?*
- *How do our chosen method and data affect the result?*
- *What insights do the results provide on how Sweden can move towards a more sustainable food consumption through market-based interventions?*

5.1 Interpretation and comparison of results

The result of this study shows that health-related costs make up the largest share of the total externalities linked to food consumption in Sweden. Among the specific dietary risk factors, low intake of wholegrains caused the largest burden, followed by high consumption of processed and red meats. This pattern aligns with data from other high-income countries, such as Norway, Belgium and Denmark [69]. In comparison, the environmental costs, although still substantial, were lower in overall magnitude. The U.N. Food Systems Summit estimates that food system-related health costs globally amount to approximately USD 11 trillion annually, while environmental costs are estimated at USD 7 trillion [21]. This suggests that health costs tend to dominate, which is in line with our findings. However, the difference between health and environmental costs in our study is even more pronounced. According to FAO, health-related costs are typically the most prominent externality in high-income countries [99], which is consistent with the results of this study. One likely explanation is the higher monetary valuation of health impacts in these contexts. The price for DALY applied in this analysis, provided by the True Price Foundation, is in line with estimates used for countries with very high Human Development Index scores [100]. This supports the interpretation that the large health-related costs observed in our results reflect established valuation practices for high-income countries, rather than an overestimation specific to the Swedish context.

Although health costs will most likely dominate regardless, there are several reasons to believe that there is an underestimation of environmental externalities in this thesis and in the overall choice of methodology. The cost of GHG emissions accounted for the largest portion of the environmental burden, and since it was monetised through SCC, the calculation is highly sensitive to ethical and methodological choices. The SCC is intended to reflect both current and future societal damages and one major source of debate surrounding the SCC concerns the discount rate, which determines how future damages are valued relative to present-day costs. Lower discount rates place greater weight on the well-being of future generations and result in significantly higher SCC estimates. Conversely, higher rates diminish the perceived value of future harms, thereby lowering the SCC. The choice of discount rate is inherently normative, reflecting ethical judgments about intergenerational equity rather than purely technical assumptions [43]. The consequences of how the different discount rates influence the total costs is analysed through the sensitivity analysis. In addition to temporal considerations, distributional equity also poses a critical ethical challenge in the construction of the SCC. Since climate change is expected to increase mortality risks through extreme weather, disease burden and food insecurity, the value assigned to human life plays a critical role in determining total damage estimates. Valuations of life and well-being are often based on WTP measurements, which are closely tied to income levels. This introduces a disparity where damages to lives in high-income countries are implicitly valued more than those in low-income countries [101]. The consequence is an ethically contentious framework that risks reinforcing global inequalities through economic modeling. This results in a SCC that underrepresents the true global cost of carbon emissions and therefore the total environmental cost estimates in this thesis. To address this imbalance, distributional considerations could be explicitly incorporated into the IAMs used to calculate the SCC. Another approach could be to apply welfare weights that adjust WTP estimates to better reflect the relative utility losses from climate change across income groups. Such adjustments would provide a more ethical and globally representative valuation of climate damages.

Perotti's assessment of the true cost of the Swiss food system [28] further suggests that the environmental externalities in this thesis likely are underestimated. The estimated annual health-related impacts for the Swiss food system were CHF 14.8 billion (approximately SEK 173 billion), while environmental and biodiversity-related impacts amounted to CHF 11.6 billion (about SEK 135 billion). Environmental costs in Perotti's study accounted for 44% of the total, whereas in this study, they represent approximately 19%. This significant difference may be partly explained by the broader scope of the Swiss study, which included 14 environmental and biodiversity-related indicators. In contrast, our analysis was limited to only 5 indicators in the final results. The categories with the highest impacts in Perotti's report include GHG emissions, cropland use and nitrogen use, which were also the largest contributors to environmental costs in this study. However, the additional impact categories not included in this assessment, such as terrestrial acidification, terrestrial ecotoxicity, and freshwater ecotoxicity, were also found to contribute substantially to the total environmental cost. This implies that a broader assessment is needed to provide a more accurate representation of the environmental costs associated with Swedish food consumption. In addition, limitations in data availability and valuation methods may have led to the exclusion of certain relevant externalities, while others were likely underestimated. For instance, the method for the cropland use category does not account for the impact of imports, meaning that costs associated with production in ecologically sensitive areas may have not been fully captured. Challenges related to imports also led to the exclusion of the water use category, resulting in the neglect of associated costs.

The results are presented both as total annual costs per food group and as per kilogram values. It is important to distinguish between these two metrics, as they serve different purposes. A high cost per kilogram does not necessarily result in a major societal cost if total consumption is low. Conversely, a low per kilogram cost may still translate into a substantial burden if the food is widely consumed. An example of this, the estimated health benefit associated with nut consumption is about SEK 182 per kilogram. However, the overall impact remains modest, as total underconsumption is relatively low. Similarly, while underconsumption of wholegrain contributes the most to the total dietary burden in Sweden, the corresponding per kilogram value is lower than for many other categories. This is because the gap between actual and recommended intake is large, spreading the cost across a broader volume of consumption shortfall. For this reason, total annual costs are more informative when identifying key contributors to societal burden, whereas per kilogram estimates provide better insight into marginal effects and are helpful when designing price-related policy interventions.

Reliable comparisons of per kilogram cost estimates across studies are limited, which makes external validation of our results challenging. Yet, our findings for red and processed meat align reasonably well with earlier estimates. Recent research commissioned by the True Animal Protein Price Coalition, based on analyses by True Price and Wageningen Economic Research, estimated the health costs related to dietary risks of red meat at approximately EUR 7.5 per kilogram [102], corresponding to about SEK 85. For processed meat, the estimated health cost was around SEK 130 per kilogram. In our study, the estimated dietary health costs were higher, with around SEK 180 per kilogram for overconsumption of red meat and about SEK 230 for processed meat. Although our figures are greater, they remain in the same order of magnitude which indicates that our methodology provides a credible result.

The same organisation also commissioned a study on the environmental costs of meat consumption, which resulted in EUR 5.7 per kilogram for beef and processed meat, EUR 4.5 for pork, and EUR 2 for chicken [102]. These correspond to approximately SEK 65, 50, and 22 respectively. Our environmental estimates were SEK 100 for beef & lamb, SEK 26 for pork, and SEK 12 for poultry. While our estimates for pork and poultry were lower, the value for beef was higher. The observed differences may reflect variations in national consumption patterns, as well as differences in methodology, including which environmental indicators were considered and how health impacts were valued. Nonetheless, the range of estimates is comparable and supports the overall robustness of our results. While there are some studies that investigate the costs associated with different dietary patterns, comprehensive assessments that quantify total health and environmental costs per specific food item remain limited, particularly when expressed in costs per kilogram. This lack of data is especially apparent in the Swedish context, where national research on food-specific external costs is still scarce. These gaps highlight the need for further studies to improve the precision.

5.2 Uncertainties

The results of this report are heavily influenced by several uncertainties related to the chosen methods. Assumptions and limitations may affect both the reliability of the data quality and how the results are interpreted. This section discusses the main sources of uncertainty and their potential impact on the results.

Cost estimates were most reliable for a subset of food items with well-established health and environmental profiles. These include beef & lamb, pork, processed meat, vegetables, fruits, milk, nuts & seeds, sugar sweetened drinks, wholegrain-products, and legumes. The results for these categories are considered more reliable due to the higher consistency in available data and scientific agreement on their impacts. For some categories, the evidence linking consumption to health outcomes is less clear. Eggs are one such example, with a scoping review highlighting mixed evidence regarding health risks [103]. The review suggests that high egg consumption may raise cholesterol levels, potentially increasing cardiovascular risk, due to the cholesterol content in eggs. However, moderate consumption is not strongly related to an increased disease risk. Given that the average egg consumption in Sweden is moderate [104], this suggests that the general population is unlikely to face significant health risks. However, some subpopulations, such as those with elevated cholesterol levels, could be at higher risk. In such cases, the absence of an assigned health cost reflects uncertainty in the underlying evidence base, rather than an assumption that there is no health impact.

There are also food categories that contribute to broader dietary risks, but cannot be directly assigned a share of the health burden. A clear example is salty snacks. As seen in GBD, high sodium intake is a major public health concern in Sweden, accounting for 10% of the total dietary disease burden [69], but is driven by the combined contribution of many foods. This makes it difficult to assign a specific share of the cost to any single food group, even if one contributes disproportionately. As a result, some categories may appear to have no health impact, even though they play a role in larger dietary patterns. Moreover, foods with variable composition, such as wholegrain-containing products, present an additional challenge. The health effect of these foods depends not only on the type of product but also on its specific contents, which introduces further uncertainty, especially when detailed content information is unavailable. In contrast to health effects, environmental impacts were estimated for all food categories, through a bottom up, LCA approach. This resulted in higher overall coverage and consistency in environmental cost estimates for the different categories.

The GBD data set provides a framework for quantifying health impacts of various dietary patterns, but there are still uncertainties related to how dietary risk factors are measured and interpreted. For instance, the dietary risk factor diet low in milk is mediated by low calcium intake, yet milk is not the sole dietary source of calcium. This raises questions about the specificity of certain food-based risk factors. A major challenge is isolating the effect of a single dietary component in the context of an entire diet, especially when individuals often consume foods in combination. Overlaps between risk factors also create potential for double counting. For instance, both diet low in fruit and diet low in vegetables are associated with similar intermediate outcomes such as elevated fasting plasma glucose, which in turn contribute to the same disease endpoints [42]. In such cases, increasing intake of either food group may reduce risk, but the combined burden may be overestimated if the effects are not fully independent. Furthermore, dietary risks are closely intertwined with other lifestyle factors such as physical activity, smoking and socioeconomic situation, all of which influence health outcomes but are not always fully accounted for in dietary models [105].

The assessment of the fish and seafood category proved to be particularly challenging due to the limitation of available data and the complexity of associated health impacts. The environmental data in our analysis mainly covered land-based systems and indicators related to agricultural practices, meaning that critical marine impacts, such as overfishing and habitat degradation, were not included. This study primarily captures carbon emissions from fishing, with over 96% of the total environmental externality costs for fish and seafood products attributed to CO_2e emissions, but likely underestimates the overall impact by excluding other significant marine environmental pressures. The difficulties in assessing health externalities contributes to the uncertainty surrounding this category. However, it is likely that fish consumption provides net health benefits, as the GBD framework identifies a health risk associated with insufficient intake of seafood-derived omega-3 fatty acids. Given that fish and seafood already show a high environmental cost in our results, the exclusion of key marine impacts suggests that the true cost may be even higher. However, the potential associated negative health cost could counterbalance some of this, making the effect on the total cost uncertain.

In addition to the limited scope of environmental impacts analysed, the main environmental uncertainties identified in this report relate to simplifications made in the monetisation of imported food products, as well as the lack of sufficient data and established methods for several impact categories. For many environmental externalities, using existing valuation methods based on the footprint indicators from the SAFAD database requires detailed information, for which data is often limited. This includes specifics such as the exact location of production, local environmental conditions and ecosystem characteristics. As a result, there is a high degree of uncertainty in categories such as new N input, new P input, blue water use, pesticide use, ammonia emissions and cropland use.

The sensitivity analysis highlights the critical role that valuation choices play in shaping the outcomes of societal cost estimates. The dominant influence of DALY valuations suggests that health externalities are the leading factor of total societal costs. The varying DALY valuations leads to large differences in the final estimates, which introduces notable uncertainties.

The asymmetric uncertainty in per-kilogram cost estimates, where upper bounds deviate more than lower ones, highlights systemic risks in relying solely on baseline values. This is especially problematic in policy contexts, where underestimating external costs can lead to inadequate or inefficient interventions. Additionally, error propagation may distort results, as initial uncertainties in data and methodology can accumulate and amplify throughout the analysis.

The sensitivity analysis underscores the need for further research to refine the accuracy of cost estimates. Improving robustness will require better data quality, enhanced methodologies, and additional measurements to reduce uncertainty and strengthen the reliability of future research.

5.3 Implications for policy interventions

The results of this study gives a rough estimate of the true cost of Swedish food consumption, as the results to a large extent are based on simplifications and assumptions. While the study shows larger trends in the impact of different foods, the results represent a range of possible hidden costs of Swedish food consumption rather than definite values. This approximation underlines the need for further research in this area that could possibly support a transformation of the food system, making it more sustainable and transparent. Despite these limitations, this section will explore how the findings, and the TCA approach more broadly, could be used to guide food policies while also highlighting key challenges, limitations and ethical considerations related to the interpretation and application of such findings.

It is important to recognise that the distinction between external costs per kilogram and total annual costs across food categories has significant implications for policy design. A policy measure with the goal of reducing societal costs should primarily focus on the foods with the highest total impact, rather than those with a high impact per kilogram. In the previous example with nuts & seeds, which have a high associated health benefit per kilogram, the total impact is relatively low compared to other foods covered in this study. Taxing or subsiding such items might therefore have minimal impact on public health and the environmental footprint as a whole. When designing policy instruments, it is therefore essential to consider both the marginal and the total effects to ensure that policies are proportionate and efficient.

This is particularly important given the fact that our method for estimating costs related to dietary risks does not account for the possibility that underconsumption or overconsumption may be concentrated within specific subgroups of the population. If the associated disease burden primarily affects a certain group rather than reflecting the national average, then applying broad policy measures may be ineffective. For instance, even when average consumption lies within the recommended range, vulnerable groups such as individuals with lower income may still experience significant underconsumption of nutritious foods and therefore bear a disproportionate share of the health burden. In such cases, population-wide interventions such as universal subsidies, may provide limited benefit. More targeted strategies directed at affected groups would likely be more effective in reducing health inequalities and improving overall outcomes.

In addition, the implementation of TCA-based policies also presents important ethical and practical considerations. In the case of animal welfare, one could argue that monetary valuation is an inadequate way to determine which policy measures should be implemented. It is important to note that most valuation methods are inherently anthropocenic and assign a value based on human preferences. Two key ethical perspectives fundamentally influence how animal welfare is incorporated into economic evaluations through WTP. A speciesist approach privileges human welfare based purely on species membership, systematically undervaluing animal life regardless of comparable capacities for suffering. In contrast, a utilitarian perspective seeks to weigh welfare impacts according to the ability to experience well-being, regardless of species, aiming for an impartial aggregation of welfare across all sentient beings.

This utilitarian foundation also underlies much of welfare economics, which forms the theoretical basis for frameworks such as TCA. From this perspective, extending economic valuation to include animal welfare is both methodologically consistent and ethically justified. As Johansson-Stenman argues, the moral relevance of sentient beings should not be limited by species membership, and many individuals intrinsically value animal well-being independently of human benefit [106]. However, in practice, the use of WTP in economic evaluations often reflects speciesist biases and fails to capture these broader ethical commitments. As a result, the interests of animals risk being underrepresented or completely excluded from calculations. In such cases, regulatory measures such as legal standards for animal treatment might be a more appropriate tool for addressing the ethical concerns of food production. These measures do not rely on market based valuations or human WTP, but instead reflect normative commitments to protect the interests of animals as moral patients.

A further challenge in designing policy measures based on TCA frameworks is the risk of overlapping with existing taxes. Since the lifecycle of food products involves multiple stages, such as primary production, processing, transportation, retail, and consumption, some environmental costs are already partially internalised through existing market instruments. For instance, emissions related to energy use in processing and transportation are often subject to fuel taxes, carbon pricing, or energy levies within the EU ETS and national taxation systems. This means that introducing an additional tax based on the total life cycle emissions or other externalities risks overlapping with costs already borne by producers or consumers. If these existing policies are not taken into account, the analysis may overestimate external costs or lead to overlapping charges that place unnecessary cost pressure on certain actors.

Composite foods, such as ready-made meals and other processed products that contain multiple different ingredients, present additional complexity to the taxation of food items since they can contain a mix of taxed and untaxed foods. Assessing the appropriate tax rate for such products could therefore prove to be difficult, especially when ingredient proportions vary between brands and the exact composition is not clearly disclosed. This uncertainty creates the risk of under- or over-taxation of products, as well as an increased administrative burden for regulators. One possible solution to this could be to apply taxes earlier in the supply chain, so that the burden falls on the manufacturer. The additional cost would then be transferred to the consumer through higher pricing, internalising the externalities without taxing the final products directly.

Another key concern is public acceptance, especially with respect to the affordability of food. Rising food prices are currently a topic for debate in Sweden, with increasing public concern about further price increases and their impact on household finances, according to Swedish media [107]. However, due to climate change, agricultural commodities might be affected concerning prices and availability. This will lead to a more volatile global food market since crop yields and supply chains will be lowered and disrupted [108]. In this context, the implementation of TCA frameworks might worsen short term affordability, particularly for low-income groups. At the same time, TCA has the potential to enhance transparency and promote more sustainable production and consumption patterns, especially if policies are designed to balance environmental, health and economic goals. To achieve this, TCA-based policies must be complemented with supplementary analysis and mechanisms that ensure social equity since this is not accounted for within the framework. A cost assessment method provided by Larsson et al. [30] demonstrates that public acceptance of a meat tax could improve when accompanied by compensatory measures such as a lowered VAT on other food items. One of the tax proposals included in the study combines excise taxes on emission-intensive and unhealthy foods, such as red meat and sugary drinks, with a VAT exemption on foods like fruits, vegetables, legumes, and wholegrains, creating a cost-neutral tax package. The results showed a significant change in consumption patterns, with a 24% decrease in the consumption of sugary drinks and a 19% decrease in the consumption of beef and lamb. Furthermore, health benefits were greater in low-income groups, suggesting that such measures could support both equity and sustainability.

In theory, the results suggest a socioeconomic net benefit associated with the consumption of certain food categories, such as fruits and nuts & seeds. These foods are linked to lower health-related external costs and generate positive health outcomes. From an economic perspective, one could argue that targeted interventions, such as providing free food vouchers could be a cost-effective policy measure. By incentivising increased consumption of nutrient-dense, low-externality foods, this proposal has the potential to reduce diet-related disease burdens and therefore lower long-term public healthcare costs. However, translating this theoretical benefit into practice presents several challenges. The current structure of the food system, including supply chains, retail pricing, and consumer behaviour, may limit the effectiveness of such interventions. Moreover, simply providing free food from a particular category does not guarantee increased intake. Food choices are influenced by an interplay of factors such as taste preferences, cultural norms, individual dietary habits and socioeconomic factors. Evidence from randomised trials suggests that while fruit and vegetable incentive programs can improve diet quality to some extent, their overall effectiveness is highly dependent on program design and contextual variables such as baseline diet quality, income level and nutrition education [109]. As a result, any policy aiming to shift consumption patterns would require a more comprehensive approach, potentially involving public health campaigns, education, and structural changes in food environments, in order to achieve meaningful and sustained behavioural change.

6
Conclusion

This study aims to assess the true cost of Swedish food consumption by accounting for environmental and health-related externalities, using a TCA approach. Monetising these externalities results in a total external cost of SEK 302 billion for the food consumption. Applying these external costs to national food expenditure suggests that the true societal cost of food consumption in Sweden could be nearly twice the current market value.

When differentiating between environmental and health externalities, the results show a clear disparity. Health-related costs account for 81% of the total external food costs, while environmental costs make up the remaining 19%. This highlights the significant role that diet plays in public health and suggests that policy efforts targeting healthier food choices may offer considerable societal returns.

A key insight from the analysis is that the magnitude of external costs varies widely across food categories. Animal-based products carry the highest external costs per kilogram due to their substantial contributions to greenhouse gas emissions, land use, and associated health risks. In contrast, plant-based foods such as legumes, vegetables, and fruits tend to have significantly lower external costs. This indicates that a dietary shift towards more plant-based consumption could yield meaningful societal cost reduction.

However, there are several uncertainties concerning the reliability of the estimated values. Limitations arise from the use of generalised food categories, absence of data, assumptions in modelling and reliance on international datasets where Sweden-specific data were unavailable. Furthermore, the monetisation of health and environmental impacts is based on economic valuation methods such as WTP, cost of damage, and value of life, all inherently normative and context-dependent. In particular, the monetary value assigned to a DALY has a major influence on the total cost estimates, as health-related impacts dominate the external costs. The sensitivity analysis shows that varying the DALY value within a reasonable range can shift the total external cost by several hundred billion SEK. All these factors introduce uncertainty to the result.

Despite these uncertainties, the results offer valuable insights. The findings provide a foundation for the design of evidence-based policy instruments, such as targeted taxes and subsidies, that could internalise external costs and guide Sweden toward a more equitable, environmentally sustainable, and health-oriented food system.

Future research could refine these estimates by incorporating more detailed consumption data, environmental and health burden metrics specific to Sweden, and a broader range of externalities. How different valuation methods for externalities affect policy outcomes and societal acceptance is also a future topic of discussion.

The large variation in external costs across food categories clearly points to structural inefficiencies in the current food system. Even if precise cost estimates could improve with better data, the overarching conclusion remains strong: current consumption patterns impose considerable costs for society, and shifting toward more sustainable and health-promoting diets can reduce these burdens.

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A
Appendix

This appendix provides a detailed overview of the estimated externality costs associated with different food categories, expressed both per kilogram and as annual costs. The purpose is to provide a detailed overview of how costs are distributed among food types.

The figures A.1 and A.2 show external costs estimated in SEK. Each category in the figures represents a typical product or group of products, combining both domestically produced and imported food covering the full scope of food consumption in Sweden. The values should be interpreted as averages based on total national consumption within each food group and do not necessarily reflect the impact of individual products or specific brands.

Cells marked with "0 SEK" indicate that data was available and assessed, but the external cost was either negligible or zero. Dashes (–) represent categories for which no reliable data could be obtained for the specific food category and externality in question.

Figure A.1 displays the per-kilogram externality costs, highlighting the marginal societal cost of consuming one additional kilogram of a given food. For high wholegrain products, where composition may vary, representative assumptions were applied: 35% wholegrain content for *bread*, 60% for *muesli & cereals*, and 100% for both *pasta* and *porridge oats*. These values are based on examples of products carrying the Keyhole label (Nyckelhålsmärket).

Figure A.2 presents the annual externality costs associated with each food category, capturing the aggregate societal impact based on actual national consumption volumes. In this figure, the total cost of underconsumption of wholegrains (approximately SEK 666 billion) could not be reliably distributed across individual product categories and is therefore excluded in total externality costs.

COST / KG	Dietary risks	Pesticide use (human toxicity)	Antibiotic Use	Air Pollution	Heavy Metal exposue	Carbon Footprint	Cropland use	Pesticide use (ecotoxicity)	New N input	Ammonia emissions	Total Cost (per kg)
Beef & lamb	181.47 SEK	0.29 SEK	1.44 SEK	1.62 SEK	0.03 SEK	67.77 SEK	23.15 SEK	0.03 SEK	7.97 SEK	1.24 SEK	**285.02 SEK**
Pork	181.47 SEK	0.36 SEK	3.78 SEK	0.64 SEK	0.03 SEK	12.30 SEK	11.52 SEK	0.03 SEK	1.92 SEK	0.49 SEK	**212.54 SEK**
Processed meat	232.59 SEK	0.34 SEK	3.47 SEK	0.66 SEK	0.00 SEK	14.48 SEK	11.48 SEK	0.03 SEK	2.11 SEK	0.50 SEK	**265.68 SEK**
Poultry	0.00 SEK	0.18 SEK	1.66 SEK	0.44 SEK	0.03 SEK	5.04 SEK	5.58 SEK	0.02 SEK	1.09 SEK	0.33 SEK	**14.36 SEK**
Fish & shellfish	0.00 SEK	0.00 SEK	3.34 SEK	0.03 SEK	0.03 SEK	15.84 SEK	2.74 SEK	0.00 SEK	0.60 SEK	0.01 SEK	**22.60 SEK**
Plant-based protein	0.00 SEK	0.18 SEK	0.00 SEK	0.01 SEK	0.00 SEK	2.63 SEK	3.63 SEK	0.02 SEK	1.23 SEK	0.01 SEK	**7.70 SEK**
Egg	0.00 SEK	0.09 SEK	0.02 SEK	0.03 SEK	0.00 SEK	2.73 SEK	3.77 SEK	0.01 SEK	0.78 SEK	0.03 SEK	**7.45 SEK**
Cheese	0.00 SEK	0.08 SEK	0.76 SEK	0.03 SEK	0.00 SEK	13.42 SEK	6.80 SEK	0.01 SEK	1.27 SEK	0.02 SEK	**22.39 SEK**
Bread - no/low wholegrain	0.00 SEK	0.06 SEK	0.00 SEK	0.37 SEK	0.04 SEK	2.52 SEK	1.97 SEK	0.00 SEK	0.48 SEK	0.02 SEK	**5.44 SEK**
Bread - high wholegrain	-31.16 SEK	0.06 SEK	0.00 SEK	0.36 SEK	0.05 SEK	2.90 SEK	1.97 SEK	0.00 SEK	0.42 SEK	0.01 SEK	**-25.38 SEK**
Flour	0.00 SEK	0.06 SEK	0.00 SEK	0.02 SEK	0.04 SEK	2.26 SEK	2.20 SEK	0.01 SEK	0.55 SEK	0.02 SEK	**5.14 SEK**
Muesli & cereals - no/low wholegrain	0.00 SEK	0.43 SEK	0.01 SEK	0.06 SEK	0.05 SEK	5.29 SEK	3.09 SEK	0.04 SEK	0.49 SEK	0.02 SEK	**9.48 SEK**
Muesli & cereals - high wholegrain	-53.42 SEK	0.31 SEK	0.00 SEK	0.06 SEK	0.00 SEK	4.10 SEK	3.66 SEK	0.03 SEK	0.44 SEK	0.02 SEK	**-44.79 SEK**
Porridge oats - no/low wholegrain	0.00 SEK	0.06 SEK	0.00 SEK	0.02 SEK	0.00 SEK	2.33 SEK	2.35 SEK	0.01 SEK	0.55 SEK	0.02 SEK	**5.34 SEK**
Porridge oats - high wholegrain	-89.03 SEK	0.05 SEK	0.00 SEK	0.02 SEK	0.03 SEK	2.58 SEK	3.99 SEK	0.00 SEK	0.48 SEK	0.02 SEK	**-81.87 SEK**
Pasta - no/low wholegrain	0.00 SEK	0.06 SEK	0.00 SEK	0.02 SEK	0.04 SEK	3.21 SEK	3.36 SEK	0.01 SEK	0.54 SEK	0.02 SEK	**7.26 SEK**
Pasta - high wholegrain	-89.03 SEK	0.05 SEK	0.00 SEK	0.02 SEK	0.00 SEK	2.37 SEK	1.72 SEK	0.00 SEK	0.43 SEK	0.02 SEK	**-84.43 SEK**
Rice & potatoes	0.00 SEK	0.04 SEK	0.01 SEK	0.01 SEK	0.03 SEK	1.35 SEK	0.69 SEK	0.00 SEK	0.13 SEK	0.00 SEK	**2.26 SEK**
Vegetables & root vegetables	-31.55 SEK	0.04 SEK	0.00 SEK	0.00 SEK	0.00 SEK	1.81 SEK	0.18 SEK	0.00 SEK	0.06 SEK	0.00 SEK	**-29.45 SEK**
Legumes	-42.02 SEK	0.18 SEK	0.00 SEK	0.01 SEK	0.05 SEK	1.46 SEK	3.78 SEK	0.02 SEK	0.40 SEK	0.00 SEK	**-36.12 SEK**
Fruit	-39.94 SEK	0.27 SEK	0.00 SEK	0.01 SEK	0.15 SEK	1.64 SEK	0.43 SEK	0.02 SEK	0.14 SEK	0.00 SEK	**-37.26 SEK**
Yoghurt & soured milk	0.00 SEK	0.01 SEK	0.12 SEK	0.01 SEK	0.00 SEK	2.54 SEK	1.26 SEK	0.00 SEK	0.23 SEK	0.00 SEK	**4.18 SEK**
Cream, sour cream, cultured cream	0.00 SEK	0.04 SEK	0.35 SEK	0.02 SEK	0.00 SEK	6.64 SEK	3.57 SEK	0.00 SEK	0.65 SEK	0.01 SEK	**11.28 SEK**
Milk	-7.25 SEK	0.01 SEK	0.12 SEK	0.01 SEK	0.03 SEK	2.37 SEK	1.21 SEK	0.00 SEK	0.22 SEK	0.00 SEK	**-3.28 SEK**
Plant-based dairy alternatives	0.00 SEK	0.03 SEK	0.00 SEK	0.00 SEK	0.00 SEK	0.91 SEK	0.31 SEK	0.00 SEK	0.03 SEK	0.00 SEK	**1.30 SEK**
Sugar-sweetened softdrinks	18.56 SEK	0.02 SEK	0.00 SEK	0.00 SEK	0.00 SEK	0.87 SEK	0.13 SEK	0.00 SEK	0.02 SEK	0.00 SEK	**19.61 SEK**
Sugar-free soft drinks	0.00 SEK	0.00 SEK	0.00 SEK	0.00 SEK	0.00 SEK	0.69 SEK	0.02 SEK	0.00 SEK	0.00 SEK	0.00 SEK	**0.72 SEK**
Beer (0 - 3.5% alcohol)	0.00 SEK	0.00 SEK	0.00 SEK	0.02 SEK	0.00 SEK	1.31 SEK	0.28 SEK	0.00 SEK	0.04 SEK	0.00 SEK	**1.66 SEK**
Juice	0.00 SEK	0.17 SEK	0.00 SEK	0.01 SEK	0.04 SEK	2.93 SEK	0.94 SEK	0.01 SEK	0.30 SEK	0.01 SEK	**4.42 SEK**
Coffee, tea & cocoa	0.00 SEK	1.07 SEK	0.00 SEK	0.10 SEK	2.82 SEK	18.16 SEK	8.43 SEK	0.09 SEK	1.78 SEK	0.06 SEK	**32.52 SEK**
Mineral water	0.00 SEK	0.00 SEK	0.00 SEK	0.00 SEK	0.00 SEK	0.63 SEK	0.00 SEK	0.00 SEK	0.00 SEK	0.00 SEK	**0.63 SEK**
Sweets & chocolate	0.00 SEK	0.61 SEK	0.12 SEK	0.02 SEK	0.11 SEK	4.43 SEK	4.32 SEK	0.05 SEK	0.47 SEK	0.01 SEK	**10.14 SEK**
Crisps & salty snacks	0.00 SEK	0.13 SEK	0.01 SEK	0.03 SEK	0.00 SEK	3.66 SEK	2.87 SEK	0.01 SEK	0.64 SEK	0.02 SEK	**7.36 SEK**
Pasteries	0.00 SEK	0.22 SEK	0.12 SEK	0.02 SEK	0.00 SEK	5.08 SEK	3.56 SEK	0.02 SEK	0.63 SEK	0.02 SEK	**9.68 SEK**
Nuts & seeds	-181.56 SEK	0.53 SEK	0.00 SEK	0.03 SEK	0.04 SEK	3.83 SEK	5.49 SEK	0.05 SEK	0.67 SEK	0.02 SEK	**-170.89 SEK**
Butter	0.00 SEK	0.09 SEK	0.83 SEK	0.47 SEK	0.00 SEK	15.79 SEK	8.36 SEK	0.01 SEK	1.53 SEK	0.03 SEK	**27.11 SEK**
Margarine	0.00 SEK	0.13 SEK	0.23 SEK	0.46 SEK	0.00 SEK	9.71 SEK	5.20 SEK	0.01 SEK	0.91 SEK	0.02 SEK	**16.68 SEK**
Vegetable oils	0.00 SEK	0.24 SEK	0.00 SEK	0.48 SEK	0.00 SEK	6.09 SEK	7.61 SEK	0.02 SEK	1.19 SEK	0.04 SEK	**15.67 SEK**

Figure A.1: Estimated externality costs (by indicator) per kilogram for aggregated food categories

COST / YEAR	Dietary risks	Pesticide use (human toxicity)	Antibiotic Use	Air Pollution	Heavy Metal exposue	Carbon Footprint	Cropland use	Pesticide use (ecotoxicity)	New N input	Ammonia emissions	Total Annual Cost
Beef & lamb	18,017,370,000 SEK	38,060,000 SEK	190,900,000 SEK	2,659,440,000 SEK	3,870,000 SEK	8,895,540,000 SEK	3,039,300,000 SEK	3,290,000 SEK	1,046,400,000 SEK	163,600,000 SEK	**34,057,770,000 SEK**
Pork	18,025,480,000 SEK	46,880,000 SEK	500,290,000 SEK	1,037,810,000 SEK	3,870,000 SEK	1,615,490,000 SEK	1,512,930,000 SEK	4,050,000 SEK	252,210,000 SEK	63,750,000 SEK	**23,062,760,000 SEK**
Processed meat	50,333,790,000 SEK	64,170,000 SEK	665,080,000 SEK	2,240,140,000 SEK	-	2,749,370,000 SEK	2,179,560,000 SEK	5,550,000 SEK	401,120,000 SEK	95,240,000 SEK	**58,734,020,000 SEK**
Poultry	-	41,580,000 SEK	375,380,000 SEK	2,076,780,000 SEK	6,620,000 SEK	1,132,610,000 SEK	1,254,760,000 SEK	3,600,000 SEK	245,500,000 SEK	74,500,000 SEK	**5,211,330,000 SEK**
Fish & shellfish	6,042,970,000 SEK	380,000 SEK	274,810,000 SEK	8,800,000 SEK	2,470,000 SEK	1,291,510,000 SEK	223,680,000 SEK	30,000 SEK	49,640,000 SEK	770,000 SEK	**7,895,060,000 SEK**
Plant-based protein	-	3,540,000 SEK	0 SEK	-	-	50,810,000 SEK	70,100,000 SEK	310,000 SEK	24,150,000 SEK	0 SEK	**148,910,000 SEK**
Egg	-	10,140,000 SEK	1,920,000 SEK	45,250,000 SEK	-	325,030,000 SEK	448,480,000 SEK	880,000 SEK	92,570,000 SEK	3,070,000 SEK	**927,340,000 SEK**
Cheese	-	14,720,000 SEK	149,850,000 SEK	111,660,000 SEK	-	2,627,790,000 SEK	1,331,830,000 SEK	1,270,000 SEK	248,180,000 SEK	4,610,000 SEK	**4,489,910,000 SEK**
Bread - no/low wholegrain	-	25,610,000 SEK	0 SEK	595,250,000 SEK	16,550,000 SEK	1,156,880,000 SEK	906,400,000 SEK	2,220,000 SEK	218,670,000 SEK	7,680,000 SEK	**2,929,260,000 SEK**
Bread - high wholegrain	*	6,010,000 SEK	0 SEK	57,940,000 SEK	4,990,000 SEK	314,430,000 SEK	214,070,000 SEK	520,000 SEK	45,610,000 SEK	1,540,000 SEK	**645,110,000 SEK**
Flour	-	4,580,000 SEK	0 SEK	15,020,000 SEK	2,960,000 SEK	178,200,000 SEK	173,580,000 SEK	400,000 SEK	42,930,000 SEK	1,540,000 SEK	**419,210,000 SEK**
Muesli & cereals - no/low wholegrain	-	7,690,000 SEK	140,000 SEK	760,000 SEK	970,000 SEK	93,430,000 SEK	54,500,000 SEK	660,000 SEK	8,050,000 SEK	0 SEK	**166,200,000 SEK**
Muesli & cereals - high wholegrain	*	1,650,000 SEK	0 SEK	230,000 SEK	-	21,610,000 SEK	19,290,000 SEK	140,000 SEK	2,680,000 SEK	0 SEK	**45,600,000 SEK**
Porridge oats - no/low wholegrain	-	50,000 SEK	0 SEK	-	-	2,220,000 SEK	2,240,000 SEK	0 SEK	0 SEK	0 SEK	**4,510,000 SEK**
Porridge oats - high wholegrain	*	730,000 SEK	0 SEK	-	360,000 SEK	35,500,000 SEK	54,820,000 SEK	60,000 SEK	6,710,000 SEK	0 SEK	**98,180,000 SEK**
Pasta - no/low wholegrain	-	6,650,000 SEK	0 SEK	30,680,000 SEK	4,030,000 SEK	345,850,000 SEK	361,090,000 SEK	570,000 SEK	59,030,000 SEK	2,300,000 SEK	**810,200,000 SEK**
Pasta - high wholegrain	*	90,000 SEK	0 SEK	-	-	5,460,000 SEK	4,000,000 SEK	10,000 SEK	1,340,000 SEK	0 SEK	**10,900,000 SEK**
Rice & potatoes	-	25,590,000 SEK	9,320,000 SEK	181,770,000 SEK	16,690,000 SEK	858,150,000 SEK	441,750,000 SEK	2,210,000 SEK	80,490,000 SEK	2,300,000 SEK	**1,618,270,000 SEK**
Vegetables & root vegetables	26,882,410,000 SEK	28,110,000 SEK	0 SEK	127,360,000 SEK	-	1,211,850,000 SEK	120,840,000 SEK	2,430,000 SEK	40,250,000 SEK	1,540,000 SEK	**28,414,790,000 SEK**
Legumes	15,253,530,000 SEK	5,280,000 SEK	0 SEK	-	1,470,000 SEK	42,140,000 SEK	109,160,000 SEK	460,000 SEK	12,070,000 SEK	0 SEK	**15,424,110,000 SEK**
Fruit	29,971,140,000 SEK	171,870,000 SEK	0 SEK	238,920,000 SEK	94,450,000 SEK	1,032,670,000 SEK	273,220,000 SEK	14,870,000 SEK	89,880,000 SEK	3,070,000 SEK	**31,890,090,000 SEK**
Yoghurt & soured milk	-	3,580,000 SEK	33,570,000 SEK	25,370,000 SEK	-	676,660,000 SEK	336,520,000 SEK	310,000 SEK	61,710,000 SEK	770,000 SEK	**1,138,490,000 SEK**
Cream, sour cream, cultured cream	-	2,590,000 SEK	24,190,000 SEK	6,460,000 SEK	-	451,170,000 SEK	242,480,000 SEK	220,000 SEK	44,270,000 SEK	770,000 SEK	**772,150,000 SEK**
Milk	4,699,640,000 SEK	8,440,000 SEK	78,900,000 SEK	249,430,000 SEK	19,320,000 SEK	1,557,420,000 SEK	791,000,000 SEK	730,000 SEK	144,890,000 SEK	3,070,000 SEK	**7,552,840,000 SEK**
Plant-based dairy alternatives	-	1,980,000 SEK	0 SEK	-	-	54,530,000 SEK	18,810,000 SEK	170,000 SEK	2,680,000 SEK	0 SEK	**78,170,000 SEK**
Sugar-sweetened softdrinks	7,423,860,000 SEK	10,350,000 SEK	0 SEK	57,720,000 SEK	-	527,860,000 SEK	80,510,000 SEK	900,000 SEK	14,760,000 SEK	770,000 SEK	**8,116,730,000 SEK**
Sugar-free soft drinks	-	990,000 SEK	0 SEK	-	-	322,600,000 SEK	11,440,000 SEK	90,000 SEK	1,340,000 SEK	0 SEK	**336,460,000 SEK**
Beer (0 - 3.5% alcohol)	-	520,000 SEK	0 SEK	1,740,000 SEK	-	152,130,000 SEK	31,930,000 SEK	50,000 SEK	5,370,000 SEK	0 SEK	**191,740,000 SEK**
Juice	-	26,620,000 SEK	0 SEK	30,610,000 SEK	6,430,000 SEK	471,740,000 SEK	152,130,000 SEK	2,300,000 SEK	48,300,000 SEK	1,540,000 SEK	**739,670,000 SEK**
Coffee, tea & cocoa	-	113,500,000 SEK	0 SEK	92,930,000 SEK	298,280,000 SEK	1,919,490,000 SEK	891,030,000 SEK	9,820,000 SEK	187,820,000 SEK	6,910,000 SEK	**3,519,780,000 SEK**
Mineral water	-	0 SEK	0 SEK	-	-	31,570,000 SEK	0 SEK	0 SEK	0 SEK	0 SEK	**31,570,000 SEK**
Sweets & chocolate	-	102,440,000 SEK	20,540,000 SEK	48,030,000 SEK	19,330,000 SEK	746,370,000 SEK	727,210,000 SEK	8,860,000 SEK	79,150,000 SEK	2,300,000 SEK	**1,754,230,000 SEK**
Crisps & salty snacks	-	7,430,000 SEK	760,000 SEK	11,150,000 SEK	-	214,390,000 SEK	168,300,000 SEK	640,000 SEK	37,560,000 SEK	1,540,000 SEK	**441,770,000 SEK**
Pasteries	-	46,500,000 SEK	26,850,000 SEK	102,520,000 SEK	-	1,096,510,000 SEK	768,270,000 SEK	4,020,000 SEK	135,500,000 SEK	3,840,000 SEK	**2,184,010,000 SEK**
Nuts & seeds	11,368,610,000 SEK	17,360,000 SEK	0 SEK	3,090,000 SEK	1,210,000 SEK	124,430,000 SEK	178,300,000 SEK	1,500,000 SEK	21,460,000 SEK	770,000 SEK	**11,716,730,000 SEK**
Butter	-	2,550,000 SEK	23,740,000 SEK	14,950,000 SEK	-	449,490,000 SEK	238,000,000 SEK	220,000 SEK	44,270,000 SEK	770,000 SEK	**773,990,000 SEK**
Margarine	-	13,800,000 SEK	25,320,000 SEK	77,030,000 SEK	-	1,046,050,000 SEK	560,120,000 SEK	1,190,000 SEK	97,930,000 SEK	2,300,000 SEK	**1,823,740,000 SEK**
Vegetable oils	-	4,550,000 SEK	0 SEK	9,770,000 SEK	-	113,300,000 SEK	141,490,000 SEK	390,000 SEK	22,810,000 SEK	770,000 SEK	**293,080,000 SEK**

Figure A.2: Estimated annual externality costs (by indicator) for aggregated food categories. Categories marked with asterisk (*) make up a total cost of SEK 666,203,600,00 due to the dietary risk of underconsumption of wholegrain

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