Emission and Dietary Policy-Interventions for Differentiated Fish products in Sweden

A QUAIDS Analysis Using Scanner Data

Tobias Nilsson

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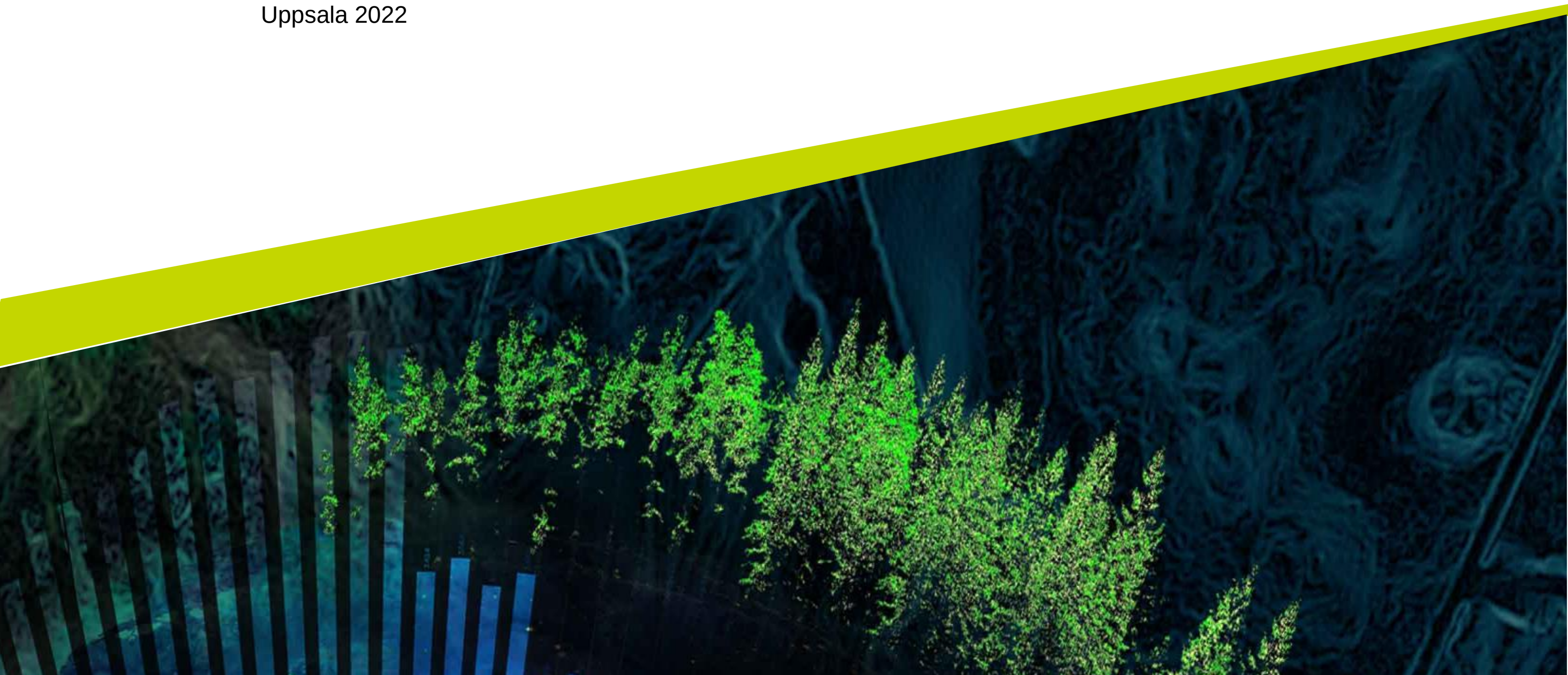

Emission and Dietary Policy-interventions for Differentiated Fish Products. A QUAIDS Analysis Using Scanner Data

Tobias Nilsson

Supervisor:	**Sarah Säll, Swedish University of Agricultural Sciences, Department of Economics**
Examiner:	Robert Hart, Swedish University of Agricultural Sciences, Department of Economics

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Swedish University of Agricultural Sciences
Faculty of Natural Resources and Agricultural Sciences
Department of Economics

Abstract

Internalizing the Social Cost of Carbon (SCC) into the price of fish is a requisite action considering Sweden's obligations under the Paris Agreement. However, bluntly taxing all seafood may conflict with Swedish authorities' dietary recommendations. Therefore, this study investigates three different tax/subsidy policy scenarios' efficacy at achieving reduced Greenhouse Gas (GHG) emissions whilst increasing compliance with public dietary recommendations. Taxes internalize products' SCC, particularly bottom trawled fish, and the subsidy at 12 percent applies to less harmful eco-labelled fish to increase dietary recommendations' fulfilment. Swedish consumers' price sensitivity is analysed using eight seafood commodity groups in a Quadratic Almost Ideal Demand System applied to 243 days of scanner data. Increased compliance with dietary recommendations comes at the expense of less GHG emissions. Nonetheless, a 'one-fell-swoop' outcome is not rendered infeasible.

Keywords: Emission-tax, Seafood, Fish, Dietary Recommendations, GHG mitigation, Sweden, QUAIDS

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Abbreviations

AIDS	Almost Ideal Demand System
ALCA	Attributional Life Cycle Assessment
ASC	Aquaculture Stewardship Council
GHG	Greenhouse Gases
LM	Lagrange Multiplier
MSC	Marine Stewardship Council
SEK	Swedish Krona
VAT	Value Added Tax
WWF	World Wide Fund for Nature
QUAIDS	Quadratic Almost Ideal Demand System

Popular science summary

IPCC estimate that food systems contribute to one-third of global Greenhouse gas (GHG) emissions. Included therein are emissions from farmed and wild seafood, which are estimated to account for 10 percent of the food systems' emissions. Internalizing the Social Cost of Carbon (SCC) into the price of fish is a requisite action considering Sweden's obligations under the Paris Agreement. However, bluntly taxing all seafood may conflict with Swedish authorities' three dietary recommendations — (i) increase overall seafood consumption by 25 percent, (ii) consume a larger share of seafood from environmentally conscious practices, and (iii) limit ingestion of fish species with higher-level ecotoxic concentrations.

Therefore, this study investigates three different tax/subsidy policy scenarios' efficacy at achieving reduced GHG emissions whilst increasing compliance with public dietary recommendations. Taxes internalize fish products' SCC, particularly bottom trawled fish, and the subsidy at 12 percent applies to fish from viable populations to increase dietary recommendations' fulfillment. Hence, the basic point for our policy designs is to calibrate the compass for socially desirable consumer behaviour whilst enhancing economic efficiency. Swedish consumers' price sensitivity is analyzed using eight seafood commodity groups in a Quadratic Almost Ideal Demand System applied to 243 days of scanner data. Our results suggest that increased compliance with dietary recommendations comes at the expense of less GHG reductions. However, a 'two-birds-one-stone' outcome is not rendered infeasible. That is, it may be feasible to induce substitution that improves compliance with dietary recommendations whilst reducing GHG emissions. Our policy scenarios positively influence two negative side effects (i.e. CO_2e emissions and overfishing), but do not consider negative side-effects such as high-grading, congestion and bycatches. Hence, they attain second-best optimums.

1. Introduction

The economic problem analyzed in this study stems from three key questions concerning seafood consumption — [1] How much should be consumed?, [2] What methods should be used in production?, and [3] Which types of seafood should be consumed?. According to Borthwick et al. (2019) average seafood consumption in Sweden currently amounts to 240 grams per person and week. However, the Swedish Food Agency (2015) recommends an increase in individual consumption amounting to 25 percent[1]. The agency's recommendations consist of three pillars — (i) increase overall seafood consumption by 25 percent, (ii) consume a larger share of seafood from environmentally conscious practices, and (iii) limit ingestion of fish species with higher-level ecotoxic concentrations. In this study, we explore consumer behaviour of seafood consumption using scanner data and find emission-tax policy interventions under which consumption is expected to shift away from products with adverse effects on health and the environment whilst reducing Greenhouse Gas (GHG) emissions.

IPCC (2019) estimates food systems contribute to one-third of global GHG emissions. Included therein are emissions from farmed and wild seafood, which are estimated to account for 10 percent of the food systems' emissions. Acknowledgement of this issue, amongst others, induced 175 world leaders to enact the Paris Agreement in 2015 (UNFCCC, 2022). Accordingly, countries are obliged to take measures to drastically cut emissions and limit global warming to 2°C (ibid). Adherence to the Paris Agreement is embedded in the Swedish Food Agency's (2015) second dietary recommendation, which also encourage a relative consumption increase of seafood from sustainable practices, e.g. Marine Stewardship Council (MSC)-, Aquaculture Stewardship Council (ASC)- and KRAV labeled products. To put this in perspective, Parker et al. (2018) present emissions intensity from different practices and find bottom trawling to be least energy efficient. Dendersen et al. (2019) explain that bottom trawling is a commonly used harvesting method whereby a weighted trawl penetrates the top-layer of the sediment and is towed over the seafloor. Beyond its GHG inferiority, Ferguson et al. (2020) and Dendersen et al. (2019) emphasize the profound negative feedback mechanisms triggered by bottom trawling. Whisking the top layer of the sediment severely disrupts the ocean's nitrogen cycle by nutrient enrichment and wipes out microbial and invertebrate interactions required for sedimenting nitrogen (i.e. denitrification). It is estimated that bioavailable nitrogen increases by 50 percent which reduces resilience towards eutrophication. This chain-reaction self-

[1] See also Ziegler (2008).

reinforce over time and ultimately creates a hypoxic uninhabitable demersal zone (ibid). Additionally, almost half of the bottom trawled fish products in our dataset are harvested in waters near Sweden, then transported to China for processing before returning to its harvest area.

The third pillar urges citizens to limit their food intake of Baltic herring, Salmonids, Big-game species and Percoids due to ecotoxic issues. Mercury, polychlorinated biphenyls (PCBs), and dioxins are given extra attention due to their adverse health effects. All of which with the potential to cause neurological disorders and disturb cognitive development, particularly amongst young children (Hellberg et al., 2012; Hughner et al., 2008; Winneke, 2011). On the other hand, consumption of seafood with safe levels of ecotoxins have been linked to health benefits such as improved neurodevelopment, healthier hearts, and balanced blood clotting (Domingo, 2016; Hellberg et al., 2012). Unfortunately, estimates of the social costs of adverse health effects from ecotoxins are not available. Hence, a health tax will not be considered in this study. However, the proposed interventions' impact on demand of ecotoxic fish will be scrutinized.

Historically, actions for fostering environmentally conscious consumption of fish and shellfish have primarily been through eco-labelling. Nonetheless, in an extensive examination of certification criteria and information campaigns by Madin & Macreadie (2015), it was found that GHG emissions are rarely used as a criterion. The authors explain that this conflicting aspect is applicable to the organization Marine Stewardship Council's (MSC) widely recognized certifications (i.e. MSC and ASC). In contrast, Swedish KRAV acknowledges climate footprint to some extent in their screening process. Beyond green consumerism, some countries have levied a carbon dioxide (CO_2) tax on fossil fuels used in farming and harvesting of fish and shellfish. Such a tax has been implemented on diesel, a commonly used fuel in commercial fishing and aquaculture, in Sweden and Norway (the Swedish Tax Agency, 2022a; Isaksen et al., 2015). However, the seafood industry in these countries can apply for tax-exemption from CO_2 taxes (the Swedish Tax Agency, 2022b; Isaksen et al., 2015). Borthwick et al. (2019) explain that seafood imported to Sweden make up 70 percent of the domestic supply and predominantly consists of Norwegian salmon. Consequently, consumption prices of seafood in Sweden does not incorporate the social cost of carbon.

This study adds to the pool of literature on climate taxes levied on food consumption, and particularly those applying demand systems to predict the impacts of climate taxes. In addition, a delimitation is set to seafood products. Previous studies have mainly focused on incorporating the social cost of carbon dioxide equivalent (CO_2e) and predicted consequential changes in demand patterns for the seafood group as a whole (see Edjabou & Smed, 2013; Forero-Cantor et al., 2020; Säll et al. 2020). By explicitly acknowledging aspects of ocean degradation in our economic policy-design, we go against the tradition of blindly focusing on GHG emissions (ibid). Although previous research utilizes empirical data (typically surveys), real life scanner data has historically only been used for descriptive demand system approaches for the North American Seafood market (see Singh et al. 2012; Sing et al. 2014; Surathkal et al. 2017). Our analysis of how policy-interventions affect Seafood consumption therefore expands the strand of literature on seafood consumption. Assessing policy-interventions' impact on dietary recommendations further adds to this study's novelty. Given the gap in the literature and outlined background, the aim of this study is to analyze if carefully designed policy interventions can achieve a 'three-birds-one-stone' outcome by inducing — increased compliance with public dietary recommendations, reduced GHG emissions, and substitution away from products causing profound ocean degradation.

Despite growing interest in implementing climate taxes on food and profound research on adverse health effects from ingesting contaminated seafood, little effort has been devoted to acknowledging both aspects simultaneously. The gap in the literature concerning substitution across different seafood product groups has also been acknowledged by Röös et al. (2021a), who encourage future research to fill the gap. Our rich scanner data from ICA Maxi provides an opportunity to predict the impact of climate taxes on demand across highly disaggregated commodity groups inside the seafood product group. Since few countries have adopted a CO_2 tax and a tax exemption can be enjoyed by the seafood industry in the two main countries supplying the Swedish seafood market (i.e. Sweden and Norway), a consumption-sided climate tax would not be considered double taxing. Given the outlined background, three policy scenarios – all aiming to reduce the sectors' GHG emissions whilst increasing compliance with dietary recommendations – are considered:

- *Scenario 1*: emission-taxation of bottom trawled fish.
- *Scenario 2:* emission-taxation of bottom trawled fish and an additional 12 percent subsidy on less harmful eco-labelled fish.
- *Scenario 3*: emission-taxation of all commodity groups, except less harmful eco-labelled fish.

This thesis' policy scenarios are analyzed using panel scanner data from one of the largest supermarkets in Sweden. First, we construct expenditure functions for each commodity group in our product range. Our system of equations is then intertwined into a Quadratic Almost Ideal Demand System (QUAIDS). Second, we use demand system estimates to calculate elasticities to discern consumers' price sensitivity and inclination to substitution. Third, we determine the magnitude of efficient taxation by consulting previous estimations of supply chain emissions. Similar to previous studies (*see* Edjabou & Smed, 2013; Forero-Cantor et al., 2020; Säll et al., 2020), we rely on climate footprints determined by Attributional Life Cycle Assessment (ALCA) from cradle to the retailer's gate. Multiplication between product specific climate footprints and Government Offices of Sweden's general social cost of carbon yields the efficient tax rate to correct for the market failure. The magnitude of the subsidy is analogous to a Value Added Tax (VAT) exemption and is introduced in an attempt to achieve a substantial increase in seafood consumption. Lastly, we utilize elasticities and price data for creating a system of demand curves that acknowledge the multi-stage budgeting process. Altogether, this approach allowed for analyzing how consumers substitute across subgroups of seafood in the event of policy intervention.

We show that solely introducing climate taxes on fish increases compliance with two dietary recommendations (i.e. consuming more of environmentally superior- and less of health bad fish), whereas a subsidy on *Eco-labelled* is required to avoid conflicting with the third recommendation of increased seafood consumption. To the best of our knowledge, a tax-subsidy scheme for fish has only been elaborated once. Marette et al. (2008) conducted a lab experiment in France and examined the impact of rewarding health-promoting sardines and penalizing mercury contaminated tuna on relative demand. The authors argue that a tax-subsidy scheme outperforms informational campaigns, in terms of achieving demand patterns for fish that comply with health objectives. Although the external validity to a Swedish context is limited due to temporal and spatial concerns, it showcases promising results from an innovative intervention worth further investigation.

This paper proceeds as follows. Section 2 provides the theoretical foundation and conceptual framework of this study, whilst Section 3 describes the properties and limitations of the data used. Section 4 outlines the methodological approach and Section 5 presents the results. Section 6 is designated for concluding remarks and discussion, followed by References and the Appendix.

2. Prologue

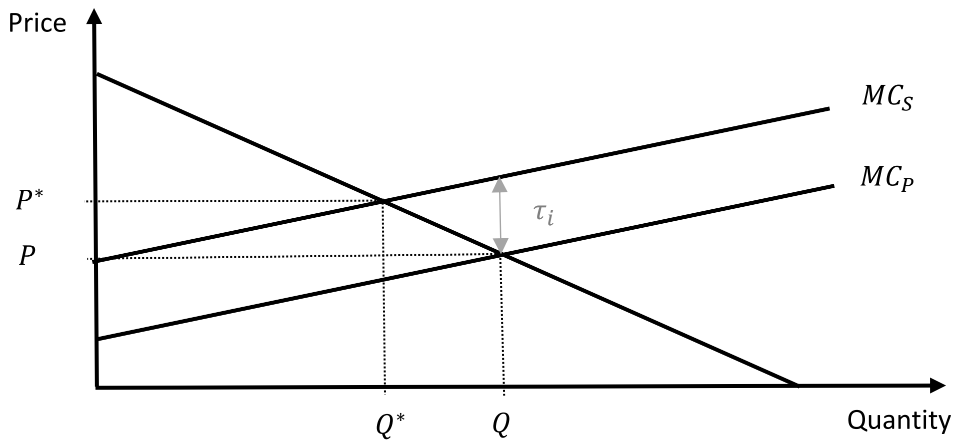
This section first explains the concept of negative externalities and the first- and second-best theorem. The preceding subsection then discerns relevant policy interventions as a means to influence consumption patterns of retail food.

2.1 Negative Environmental Externalities

The economic definition of a negative environmental externality, as suggested by Kolstad (2011, p.87), emphasizes the causal unwanted harm on one actor from another actor's actions. Moreover, the author stresses that the harm must be unpermitted by the harmed party who receives no compensation for its decreased utility or production losses. Waldo et al. (2016) explain that seafood production is associated with a wide range of negative environmental externalities. The most renowned are said to be open-access externality (i.e. one vessel's harvest involuntarily affected by another's), bycatch, and CO_2e emissions. The latter is explicitly emphasized in this study, together with overfishing. Kolstad (2011) explains that in the absence of government intervention, the market price fails to signal the social cost of pollution. Instead, the author suggests implementing taxes to correct for the price of pollution, i.e. Pigouvian taxes. By means of the author's definition of a negative environmental externality, seafood production imposes uncompensated costs on global citizens in the absence of a Pigouvian tax. In turn, goods with unpriced negative externalities exhibit excess consumption (ibid; Edjabou & Smed, 2013).

The market failure of negative environmental externalities and the impact of a correcting tax is illustrated in *Figure 1*. The conceptual illustration and associated explanations of Kolstad (2011) are tailored for our case. Two scenarios are considered. First, the inefficient equilibrium is found at the intersection between the private marginal cost (MC_P) and Marginal Benefits (MB). The inefficiency stems from excess consumption of seafood ($Q_m > Q^*$) due to the failure of incorporating the product's social cost in the market price ($P_m < P^*$). Supply chain activities responsible for negative externalities from seafood consumption include harvesting, cooling, processing, transportation, and waste management. Second, internalizing the social costs causes a parallel upward shift of the MC_P curve. The internalization requires determining the climate footprint and taxing the commodity accordingly (τ_i), i.e. introducing a Pigouvian tax (Pigou, 1957). This induces the efficient equilibrium found at the intersection between the Marginal Social Cost (MC_S) and MB. At this point, seafood consumers face a market price that incorporates external harm (P^*) and consume a socially optimal amount (Q^*).

Figure 1. Conceptual illustration of negative externalities.



Source: Own illustration using Kolstad (2011, p.250).

Boadway & Bruce (1984) explain that the existence of negative environmental externalities such as open-access overfishing (due to lack of property rights) or jointness between one's consumption and others' disutility (due to excess consumption) violates Pareto optimal allocations. The authors argue that corrective interventions to tackle several externalities can achieve one of two optimums. The first-best optimum is secured by resolving all market failures simultaneously and is typically achieved through multiple redistributive policies. If a single policy, e.g. set of taxes/subsidies, positively influence several (but not all) externalities, the attained outcome is called a second-best optimum.

Perman et al. (2011) recognize the precarity of correcting for emissions from production through the lens of the second-best theorem. It is suggested by the authors to conduct a thorough analysis of — elasticity of demand, firms' margins, and harm from consumption, in order to determine whether an intervention yields positive net benefits. In addition, the authors argue that the second-best policy should acknowledge which market failures cannot be corrected. According to Kolstad (2011), public bads such as GHG emissions is one market failure which cannot be corrected through agreements. The impediments to reaching agreements to ban emissions include failure to determine the point sources of emitters and diffusion of harm, as well as achieving international coordination.

2.2 Food Policies

Fundamental to promoting sustainable retail food consumption is influencing consumption patterns via policy instruments with adequate acceptance. Lack of acceptance can, according to Röös et al. (2021b), be a substantial hurdle to policy implementations and a climate tax on food is put forward as especially controversial. In fact, somewhat less than fifty percent of Swedes consider a climate tax on beef undesirable (Andersson et al., 2020), whilst the acceptance of a climate tax on seafood has not yet been examined. Röös et al. (2021b) catalog food consumption policies into three categories — informational campaigns, economic instruments, and legal regulations. All three are currently pursued in Sweden, albeit with varying impacts on demand and acceptance amongst Swedes.

Informational campaigns, as suggested by Röös et al. (2021b), include tools such as positive eco-labelling and consumer guides. Positive eco-labels are printed on certified producers' packages to signal compliance with sustainability criteria. The authors point out two weaknesses and one strength with this policy instrument. On one hand, it fails to achieve lasting impacts on demand as well as substitution away from product categories with high climate footprints. On the other hand, it signals which substitutes within the product category are superior in terms of certain sustainability criteria. In Sweden, there are three positive eco-labels for fish and shellfish — MSC, ASC, and KRAV (the Swedish Food Agency, 2022). However, Röös et al. (2021b) explains that these certifications are voluntary. Furthermore, fishermen and aquaculture farmers must apply for and cover the costs for the certification (MSC, 2022; ASC, 2017; KRAV, 2021). A well renowned consumer guide for seafood consumption is World Wide Fund for Nature's (WWF) Fish Guide. WWF (2022) base their recommendations on estimations of stock resilience and negative externalities from different harvesting and farming practices. Concerning production practices, the organization emphasizes the negative impacts from bottom trawling. Lindahl & Jonell (2020) explain that there are signals of WWF's guide being an accepted intervention with impacts on food retailers' assortments. Since the organization's recommendations mirror those of the Swedish Food Agency, this study's proposed interventions may have sufficient acceptance amongst food retailers.

Economic instruments can be implemented to incentivize or penalize isolated groups of commodities. Applied to a Swedish context, Röös et al. (2021b) explains that the most commonly used economic instrument for food products is intervention of the VAT. An increase in VAT for specific products is called an excise tax, whereas a decrease in VAT is analogous to a subsidy. Although excise taxes can correct for environmental damage, they have not yet been implemented on Swedish food consumption. As for now, the VAT rate on food in Sweden is twelve percent, which is lower than the status quo VAT at 25 percent (the Swedish Tax Agency, 2022c).

Legal regulations are found on the highest rung of the policy maker's ladder. Röös et al. (2021b) explain that Sweden currently has no laws aiming to promote consumption patterns of sustainable food. Potential legislative interventions, as suggested by the authors, are restricting availability- and marketing of food with adverse impacts on the environment or human health. Although legislative restrictions apply to sales of tobacco and alcohol, no analogous intervention has been applied to Swedish food consumption. Despite a prevalent aversion towards banning products with high environmental impact, the authors explain it is not unaccustomed. For example, filament light bulbs have been banned in the European Union due to its environmental inferiority, relative to substitute light bulbs. Similarly, the authors mention that regulations targeting unhealthy food are underway in Scotland. To date, no legislative mandate on declaration of ecotoxin concentration is required on product packaging. Instead, it is expected that consumers consult public dietary recommendations.

3. Data

This section is designated for describing the scanner data on seafood, as well as the data processing prior to estimations. Subsequently, descriptive statistics, climate footprints, and data limitations are spelled out.

The supermarket ICA Maxi Nacka provided the scanner panel data used in this study. Sampling was conducted by the supermarket during August 1st 2020 to March 31st 2021 (i.e. 243 days). The raw data contains daily information about products' – weight, quantity sold and retail prices excluding VAT. The location of the supermarket is in Nacka, Sweden, and its clientele is expected to be dominated by middle- to high income households.

3.1 Data Processing

The first step of the data analysis involved performing several data interventions in the raw data. First, outliers had to be identified and excluded from the analysis. This applied to all products with fewer than three observations and profound seasonality in sales. Seasonality was particularly strong for pickled herring and oysters. Weights of sold oysters were not reported and they were primarily sold during New Year's Eve. At dates adjacent to major holidays, pickled herring showed clear spikes in sales and prices as low as 1 SEK/kg. Therefore, they were omitted too. Whenever omitted products were included, heteroscedasticity issues incremented in subsequent estimation. Unfortunately, applying moving averages to deviating demand for these products proved insufficient. Given the product-specific rationale for exclusion and heteroscedasticity issues faced, omitting these products was considered the most sensible approach.

Data on products' harvest method, eco-label, degree of processing (e.g. breaded or not) and species classification was important for this study and had to be complemented[2]. This complementary data was gathered from visual inspection of packages and used for sorting the remaining 471 products into groups of products. Once products were sorted, the 12 percent VAT was added to reported values. Dividing daily sales with corresponding quantity sold for each product allowed for retrieving their daily prices per kilogram. Ultimately, products were aggregated into eight different aggregations of products as shown in *Table A1* in the Appendix. In brevity, *Fish* includes all fresh and frozen fish sold whole or as filets. *Canned*

[2] European regulation (1379/2013/EC) requires fish and shellfish products from capture fisheries to declare harvesting method.

Seafood aggregates all canned and jarred seafood. *Breaded Seafood* contains all seafood products diluted with or covered in cereal. *Shellfish* aggregates fresh and frozen crustaceans and squid. *Eco-labelled* products include all fish certified with MSC, ASC, or KRAV and caught with less harmful methods. *Bottom trawled* fish isolate all filets of- or whole fish caught with bottom trawling. *Conventionally produced* products include non-certified fish caught with less harmful methods. *Ecotoxic* fish include Baltic herring, Percoids, and Big-game species with concentrations of ecotoxins above recommended levels, caught with less harmful methods (see *Figure 2*).

Figure 2. Fish species the Swedish Food Agency recommends consuming less of.

BALTIC HERRING	PERCOIDS:	SALMONIDS:	BIG-GAME SPECIES:
• **Fishing area**: the Baltic Sea (FAO27). • **Species**: Baltic Herring • **Ecotoxin**: PCBs and dioxins.	• **Fishing area**: Swedish lakes (FAO5). • **Species**: Perch, Zander, Pike, and Burbot. • **Ecotoxin**: Mercury.	• **Fishing area**: Vänern, Vättern (FAO5) and the Baltic sea (FAO27). • **Species**: Salmon and Trout. • **Fishing area**: Vänern and Vättern (FAO 5). • **Species**: European Whitefish. • **Fishing area**: Vättern (FAO 5). • **Species**: Arctic char. • **Ecotoxin**: PCBs and dioxins.	• **Fishing area**: N/A. • **Species:** Tuna, Swordfish, Hallibut, Shark, and Rays. • **Ecotoxin**: Mercury.

Source: Own illustration using recommendations from the Swedish Food Agency (2015). Notes: The recommendation for Tuna is only applicable to fresh or frozen cuts since a different species is used in e.g. canned tuna.

After products had been aggregated into commodity groups, some seasonality issues remained and caused issues with Lagrange Multiplier (LM) heterogeneity tests in subsequent estimation. This was dealt with by replacing price and quantity sold for 23rd-25th December and 30th December to 1st January with seven days moving averages. By doing so, the underlying trend is better exposed and seasonal noise removed.

3.2 Descriptive Statistics

Table 1 summarizes average sold quantities and prices for commodity groups of seafood, on a daily basis. Among all, *Fish* is the most sold type of seafood with average daily sales amounting to 377kg per day. *Shellfish* is the second most sold seafood with daily average sales of 182kg per day. Average sales of *Canned*- and *Breaded Seafood* is lower and amounts to circa 45kg per day.

Average prices per kilogram show that *Shellfish* is the most expensive subgroup of seafood at 220 SEK/kg, followed by *Fish* at 208 SEK/kg. *Canned-*, and *Breaded Seafood* show lower average prices at 110 and 94 SEK/kg, respectively

Table 1. Descriptive statistics for disaggregation of seafood 2020-2021

Variable	Mean	Min.	Max	Observations
Q_{Fish}	377.00 (134.83)	176.00	1,092.71	243
$Q_{Canned\ Seafood}$	49.26 (12.39)	22.34	88.42	243
$Q_{Breaded\ Seafood}$	42.05 (11.15)	17.07	90.77	243
$Q_{Shellfish}$	182.09 (133.57)	32.43	844.84	243
P_{Fish}	207.89 (31.60)	134.46	284.46	243
$P_{Canned\ Seafood}$	109.83 (8.48)	81.70	138.66	243
$P_{Breaded\ Seafood}$	94.36 (13.88)	74.20	172.44	243
$P_{Shellfish}$	220.09 (51.65)	105.60	383.10	243

Source: Own table using data from ICA MAXI.

Notes: Numerical values show daily averages of quantities sold and prices, with standard deviations reported in parentheses.

Table 2 presents the descriptive statistics for different commodity groups of *Fish*. *Conventionally produced* fish is the most sold group of products with average daily sales at 255kg per day. The second most sold category of sold fish is *Eco-labelled* fish with daily sales of 64kg per day. The explanation for the substantial difference between these two average daily sales is twofold. First, the most consumed fish Salmon is packaged in different sizes depending on if it is eco-labelled or conventional. Eco-labelled salmon is typically smoked and sold in packages of circa 200 grams. Conventional salmon is typically fresh and sold whole in packages of circa 1.5kg or in relatively large cuts. Second, data on whether salmon sold over the counter is certified or not is difficult to access. After visiting the supermarket, we realized that most fish sold over the counter do not have eco-labels visible next to them. Furthermore, suppliers of fish sold over the counter vary frequently. Therefore, it was not feasible to determine which weeks salmon sold over the counter were eco-labelled and not. Consequently, most salmon sold over the counter were classified as conventional. *Bottom trawled* fish display average sales of 42 kg per day, and *Ecotoxic* products sell 16kg per day on average. It is worth noting that the latter two aggregations consist of both eco-labelled and conventional products. Furthermore, 40 percent of the products in the bottom trawled commodity group are harvested in waters near Sweden (i.e. the Northeastern Atlantic, FAO 27), then transported to China for processing before returning to its harvest area.

Regarding prices of different types of fish, *Ecotoxic* fish is the most expensive with an average price at 285 SEK/kg. The high price was primarily driven by the Big-game species Tuna and Halibut. The second most expensive fish is *Conventionally produced* fish with an average price of 227 SEK/kg. Its relatively high price originates from a large share of premium products sold over the counter. The third most expensive is *Bottom trawled* fish with an average price of 172 SEK/kg, whereas *Eco-labelled* fish is the cheapest with an average price of 169 SEK/kg.

Table 2. Descriptive statistics for disaggregation of seafood 2020-2021

Variable	Mean	Min.	Max	Observations
$Q_{Eco\text{-}labelled}$	63.93 (41.11)	14.32	314.08	243
$Q_{Bottom\ trawled}$	42.15 (15.16)	14.06	133.06	243
$Q_{Conventionally\ produced}$	255.34 (112.74)	79.72	656.61	243
$Q_{Ecotoxic}$	15.59 (10.81)	1.73	80.95	243
$P_{Eco\text{-}labelled}$	169.40 (22.26)	97.41	245.00	243
$P_{Bottom\ trawled}$	172.34 (11.28)	120.52	193.76	243
$P_{Conventionally\ produced}$	226.54 (44.97)	126.00	342.49	243
$P_{Ecotoxic}$	284.65 (80.93)	103.05	502.80	243

Source: Own table using data from ICA MAXI.
Notes: Numerical values show daily averages of quantities sold and prices, with standard deviations reported in parentheses.

The reasons why the numerical distance from minimum to maximum quantities and prices vary substantially for each commodity group is due to endogenous and exogenous factors. One endogenous factor is that occasional sales occurred during the sampling period. Exogenous factors include holiday consumption patterns and preferences for shopping during weekends, over weekdays. Similarly, the sizable standard deviations are consequences of fluctuating demand. Admittedly, we were unable to determine the degree of price variation stemming from endogenous and exogenous factors, respectively. For instance, if the sizable standard errors are primarily attributable to endogenous factors such as sales at times of high demand, it would cause systematic overestimations of subsequent price elasticities. Nonetheless, our smoothing during times of deviant demand as well as omission of highly seasonal products ensured that the price trend is consistent throughout the time period.

3.3 Climate Footprints for Seafood Products

Procuring the climate footprints for each commodity group of seafood is crucial for two reasons. First, it is necessary for calculating the magnitude of the Pigouvian taxes. Second, it exposes how GHG emissions change from tax-induced substitution across commodity groups. For the latter purpose, the climate footprints from Carbon cloud (2022) allows for connecting climate footprint to all commodity groups, whilst including climate footprints for the most sold species. Additionally, only using one source for climate footprints ensures a consistent procedure for attaining average climate footprints. Therefore, the baseline scenario solely uses climate footprints from Carbon Cloud (2022). Admittedly, this comes at the cost of neglecting the climate footprints for seven species found in Moberg et al. (2019). Inclusion of these species primarily affect the climate footprint for the commodity groups *Eco-labelled* and *Conventionally produced* fish[3]. Incorporating climate footprints for species included in the baseline scenario further increases the climate footprints for the other commodity groups since they are estimated to be higher in Moberg et al. (2019). Hence, the results' sensitivity to this exclusion is elaborated in this study's sensitivity analysis.

Table 3 provides an overview of seafood products' climate footprints from cradle to Swedish retailers' gate (Carbon Cloud, 2022; Moberg et al., 2019). Tabulated values express each commodity group's CO_2e per kilogram of edible product. Average climate footprints are weighted according to average sales of each subsumed species.

Regarding the baseline scenario, it is found that *Seafood* is associated with 5.20 kg CO_2e per kilogram of edible product. Within this group, *Shellfish* displays the highest climate footprint at 8.68 kg CO_2e per kilogram of edible product. The second highest climate footprint is found for *Breaded Seafood* at 5.39 kg CO_2e per kilogram of edible product. *Canned Seafood* and *Fish* are associated with circa 4 kg CO_2e per kilogram of edible product, respectively. Within the Fish product group, *Bottom Trawled* fish is the product group associated with the highest climate footprint at 7.87 kg CO_2e per kilogram of edible product. The average climate footprint of species included in the *Eco-labelled* and *Conventionally produced* amounts to circa 3 kg CO_2e per kilogram of edible product. Unfortunately we were unable to find climate footprints for species subsumed in *Ecotoxic* fish. Since both eco-labelled and conventionally produced products are represented in this aggregation, we apply the average climate footprints for conventionally produced- and eco-labelled fish as a proxy. Hence, its climate footprint may be less accurate than the climate footprints for the other third level commodity groups.

[3] See Table A2 in the Appendix for a full presentation of species included for average climate footprints in the reference- and sensitivity analysis scenarios, respectively.

Table 3. Climate footprints of Seafood products

Product	CO_2e/kg edible product (Carbon Cloud, 2022)	CO_2e/kg edible product (Moberg et al., 2019; Carbon Cloud 2022)
Average: Seafood	5.20	6.49
Average: Fish	3.54	4.87
Average: Canned Seafood	4.90	4.90
Average: Breaded Seafood	5.39	6.13
Average: Shellfish	8.68	10.37
Average: Eco-labelled	3.10	4.49
Average: Bottom trawled	7.87	8.56
Average: Conventionally produced	2.96	4.38
Average: Ecotoxic	3.03	4.44

Source: Own calculations using estimates from Carbon Cloud (2022) and Moberg et al. (2019). *Notes:* R22-emissions from cooling medium leakage are omitted from Moberg et al. (2019) since this cooling medium has been phased out. Both sources calculate climate footprints using GWP_{100} and ALCA.

3.4 Data Limitations

Perhaps the most overarching data limitation is concerned with external validity since the scanner data comes from a single supermarket. Ideally, one would use scanner data from supermarkets in rural, as well as differently populated regions for the purpose of expanding this study's result to regional or nationwide implications. Moreover, ICA Maxi Nacka is located in the metropolitan suburb of Nacka in Stockholm with a relatively strong representation of middle- to high income clientele. Another limitation with our approach is that we solely rely on panel scanner data. Ideally one should complement this data with household data for households shopping at this particular store. Not having household data implies that there is a risk of misinterpreting an increase in number of shoppers as if households are getting richer. Regarding the complementary product information, we realized that seafood sold over the counter frequently changes suppliers. As suppliers change, so could the products' status of eco-labelling. Therefore, a few products had to be classified as conventionally produced throughout the 243 days.

Retrieving climate footprints for species subsumed in the commodity group *Ecotoxic* fish proved particularly challenging, and a proxy had to be used to circumvent this limitation. In addition, we only found the climate footprint for one subsumed product of *Canned Seafood* (i.e. tuna). Furthermore, since *Breaded Seafood* is typically made from Cod, Salmon and Plaice, its climate footprint is approximated as a weighted average of these products' climate footprint. Ideally, its climate footprint should reflect the seafood/cereal ratio, which approximately is 80:20 in our sample, as well as the breading process. Lastly, during the completion of this study, inflation rates increased sharply which undermines the intertemporal validity. Despite these limitations, well grounded estimates for the most consumed species – Salmon, Herring, Cod, Shrimp, and canned tuna – are utilized.

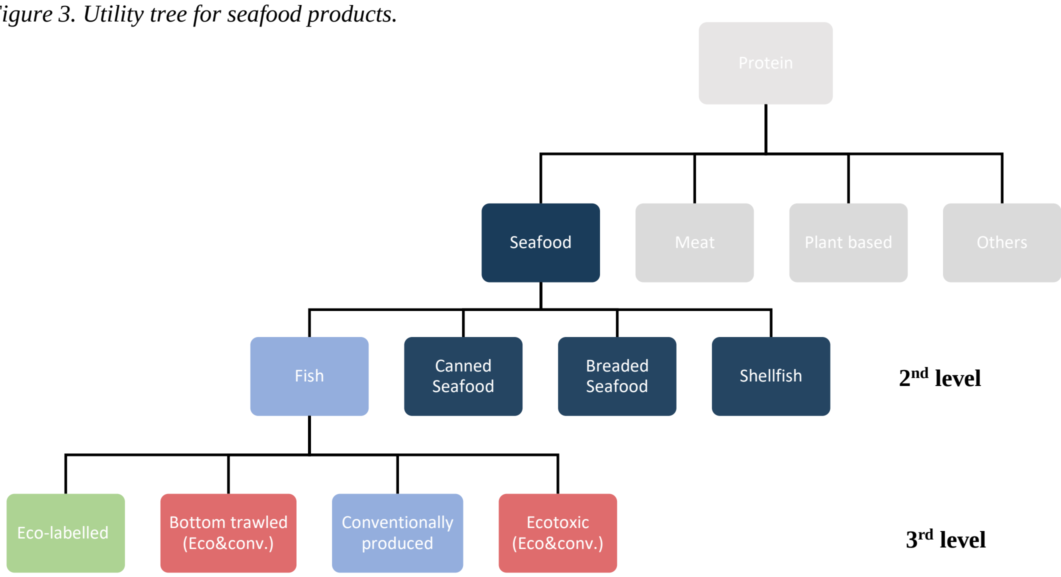
4. Method

This section explains the methodological approach applied in our analysis. The first subsection outlines the design of the demand-system and the associated multi-stage budgeting process. Following this, we provide our rationale for choosing the Quadratic Almost Ideal Demand System (QUAIDS) as our vehicle for elasticity calculations. Emphasis is placed on key steps in calculations and underlying assumptions. Moreover, the procedures for determining Pigouvian taxes and policy-induced impact on demand and GHG emissions are covered in the last subsection.

4.1 Two-stage Demand System

Designing an adequate demand system requires relying on method specific assumptions, as well as assumptions concerning consumer preferences. Fundamental to multistage budgeting are the assumptions of weak separability and utility maximizing consumers (Edgerton, 1997). Assuming weak separability allows for aggregating products to separate product groups and ensures demand across groups behave similarly, following a price alteration (ibid). The author utilizes Swedish consumption data and presents the sequential decision process they found most adequate. First, the consumer settles upon its budget share devoted to retail food. Second, the representative consumer distributes budget shares across different aggregated groups of protein, beverages, vegetabilia, and miscellaneous, based on relative price considerations. Third, preceding re-allocation of budget shares takes place across aggregations of protein products such as seafood, meat, plant-based, and other sources of protein. Subsequent allocation of budget shares across differentiated Seafood products is illustrated as coloured commodity groups in the utility tree in *Figure 3*.

Figure 3. Utility tree for seafood products.



Source: Own illustration, using Edgerton (1997), Bronnman et al. (2019), and Säll et al. (2020). *Notes:* Individual products are only subsumed in one commodity group at each level.

The disaggregation of seafood into commodity groups hinges upon own assumptions concerning consumer preferences and is particularly inspired by Bronnman et al. (2019). At the second level, disaggregation yields groups of *Fish*, *Canned Seafood*, *Breaded Seafood*, and *Shellfish*. Since *Fish* is the only commodity group associated with dual societal costs (i.e. adverse effects on health and environment) it is given extra attention on a third level. The third subsystem reflects the consumers' choice between *Eco-labelled* and *Conventionally produced* fish. For the purpose of this study, two additional product groups are isolated. That is, *Bottom trawled* and *Ecotoxic fish*. It is worth noting that these two commodity groups are isolated due to their adverse effects and not as an attempt to reflect consumer preferences. In a stated preference study by Bronnman (2016, p.74), the author finds that most German seafood consumers claim they consider harvesting method when purchasing fish. However, no analogous stated preference study for Swedish consumers' demand structure exist to date.

4.2 Quadratic Almost Ideal Demand System

For the purpose of analyzing the potential impact of an indirect tax on consumer behaviour there are several methods to choose between. Barnett & Seck (2008) explain that two of the most prominent demand system models are the Rotterdam model and the Almost Ideal Demand System (AIDS). In a performance test, the authors analyzed to what extent these models achieve to recover true elasticities. Model estimations with subsequent Monte Carlo simulation were preceded with a comparison to true elasticities. Overall, both models perform seemingly well whilst AIDS' standard errors are smaller in all considered cases, relative to those of the Rotterdam. Hence, AIDS' precision may be somewhat superior to the Rotterdam model.

In the seminal article on the AIDS by Deaton & Muellbauer (1980), the authors explain the coherence of their proposed demand system. In terms of generalizability, it is similar to the Translog and Rotterdam demand systems but nests properties that in some instances only are found in one of the two. Nested properties of the AIDS are first-order approximation which is applicable to all demand systems, ideal aggregation of consumption patterns, functional forms matching real market data, user-friendliness, and compatibility for testing homogeneity and symmetry restrictions.

Despite AIDS being strong in terms of precision and coherence, it is weak in one important aspect. That is, its assumption of linear Engel curves that are monotonic in utility. Banks et al (1997) shows that a violation of this assumption causes biased estimates in the welfare analysis. In particular, welfare losses from indirect taxes for households with above and below the average income level will typically be underestimated. Empirical evidence presented by the authors reveal that patterns of total expenditure indeed fail to satisfy this assumption. To circumvent this issue, the authors develop the model by raising the logarithmized income term to the second power. Allowing for flexibility in the Engel curve veered the authors to replace the PIGLOG expenditure function in the AIDS with a proposed indirect utility function. Notwithstanding these deviations, the QUAIDS intertwines both AIDS and Jorgenson's Translog models and retain flexibility in price responses. The logarithmized quadratic specification also allows each product to be a necessity and luxurious at different income levels. Altogether, the QUAIDS is consistent with microeconomic demand theory and observations on consumer behaviour. The key elements of the QUAIDS's model specification (see Banks et al, 1997; Deaton & Muellbauer, 1980) are:

The **expenditure share functions** (s_i) for each aggregation of products ($i = 1, \dots, n$):

$$s_i = \alpha_i + \sum_{j=1}^{n} \gamma_{ij}\, ln[p_j] + \beta_i(ln[X] - ln[P]) + \frac{\mu_i}{Q}(ln[X] - ln[P])^2 \qquad (1)$$

These are also called the EQUAIDS demand functions, where each expenditure share is regressed on the logarithmic prices of product groups ($j = 1, \dots, m$), total expenditures ($X = \sum_{i=1}^{n} p_i q_i$), an aggregated price index (P), and a Cobb-Douglas aggregator (Q).

The **aggregated price index**, adapted for the quadratic specifications, is

$$ln[P] = \alpha_0 + \sum_{i=1}^{n} \alpha_i ln[p_i] + \frac{1}{2}\sum_{i=1}^{n}\sum_{j=1}^{n} \gamma_{ij} ln[p_i] ln[p_j] \quad (2)$$

The **Cobb-Douglas price aggregator** is defined as

$$Q = \prod_{i=1}^{n} p_i^{\beta_i} \quad (3)$$

In our application of the QUAIDS, consumption variables are lagged, as suggested by for example Alessie & Kapteyn (1991) and utilized by Säll & Gren (2015).

Concerning the parameters in the first equation, the effect of changes in p_j (ceteris paribus) on the ith budget share is working through each of the gammas (γ_{ij}). Changes in real expenditure is captured by the betas (β_i), and a positive (negative) sign suggests it is a luxury (necessary) good. In addition, there are parameters for the intercept (α_0) and the quadratic term (μ_i). Adherence to QUAIDS properties requires that five parametric restrictions are fulfilled. The first three parametric restrictions are imposed by construction and require budget shares to sum to one $\sum_{i=1}^{n} \alpha_i = 1$ as well as $\sum_{i=1}^{n} \beta_i = \sum_{i=1}^{n} \mu_i = 0$. Edgerton (1997) explains that these **adding-up restrictions** allow for eliminating one expenditure share function on each level of the utility tree to avoid singularity problems. The fourth restriction requires **homogeneity** $\sum_{i=1}^{n} \gamma_i = 0$ for the parameters displaying how budget shares change as prices change and is satisfied by construction. The fifth restriction require **Slutsky symmetry** $\gamma_{ij} = \gamma_{ji}$. This imply that changes in the price of product group i should cause a marginal effect on the budget share of product group j proportional to the marginal effect of a price change of good j on budget shares of good i. Moreover, estimation requires assuming weak intertemporal separability. That is, the budget share (of the consumer's total budget) designated for food consumption is assumed to be static.

Following the finding of distinct seasonal fluctuation in demand of fish by Capps & Lambgrets (1991) and Johnston et al. (1998), there has been a trend of adding seasonal control dummies to the specification (see e.g. Singh et al., 2014; Bronnman, 2016). We consider such a respecification inadequate to our study since we lack data on an entire year and since a large share of seafood in our data is frozen and therefore insensitive to seasonal harvesting cycles.

4.3 Price- and Income Elasticities

Estimates concerning how income or price alterations influence consumption patterns are obtained by analyzing income and price elasticities. For this purpose, we calculate Income-, Marshallian- and Hicksian elasticities as specified in

Edgerton (1997), albeit with the notation found in Säll et al. (2020):
$$\varepsilon_i^I = 1 + \frac{\beta_i}{s_i} \tag{4}$$

$$\varepsilon_{ij}^M = [\frac{\gamma_{ij}-\beta_i s_j}{s_j}] - \delta_{ij} \tag{5}$$

$$\varepsilon_{ij}^H = \varepsilon_{ij}^M + s_j \varepsilon_i^I \tag{6}$$

Superscripts code for income- (I), Marshallian- (M), and Hicksian (H) elasticities. All parameters originate from estimation of equation (1), except for the subtractive Kronecker delta. This parameter is introduced in equation (5) and takes on value one if $i = j$, and zero if $i \neq j$. Once elasticities are obtained, it must be tested if the requirement of homogeneity of degree zero, i.e. $\varepsilon_i^I + \sum_{i=1}^n \varepsilon_{ij}^M = 0$, is satisfied.

Since the elasticities presented above neglects the multi-stage budgeting process, further calculations are necessary to arrive at final elasticities (Edgerton, 1997):

$$\varepsilon_i^{I*} = \varepsilon_i^I \varepsilon_r^I \varepsilon_a^I \tag{7}$$

$$\varepsilon_{ij}^{M*} = \delta_{ab}\delta_{ru}\varepsilon_{ij}^H + \delta_{ab} s_j \varepsilon_i^I \varepsilon_{ru}^H + s_j s_r \varepsilon_i^I \varepsilon_r^I \varepsilon_{ab}^M \tag{8}$$

Subscripts code for product groups on each level in the demand system. The first level is denoted ($a \equiv b = 1, \ldots, c$), the second level ($r \equiv u = 1, \ldots, k$), and the third level ($i \equiv j = 1, \ldots, n$). As the analysis transitions upwards in the demand system, lower-level factors drop out from the equations. Since final compensated elasticities do not account for income effects, we emphasize final uncompensated elasticities instead.

4.4 Tax and Subsidy Induced Changes in Demand and GHG Emissions

Different policy scenarios' impact on demand and GHG emissions are simulated by first creating a system of demand curves. For simplicity, we consider linear demand functions of own-, and cross price elasticities. This system is built to reflect consumers' willingness to pay contingent on price- and income elasticities, where initial consumption (q_i^0) and new quantity consumed (q_i^1) are discerned through:
$$q_i = k_{ij} p_i + m_i + \Delta h_i \tag{9}$$

The negative slope of a product group's demand curve ($k_{ij} = \frac{\Delta q_i}{\Delta p_j}$) is retrieved by re-arranging the final Marshallian demand $\varepsilon_{ij}^{M*} = \frac{\Delta q_i}{\Delta p_j}\frac{p_j^0}{q_i^0}$, when $i = j$. A commodity group's price is represented by the term p_i. Before policy interventions are introduced, the initial demand curve's intercept is merely m_i and the sum of intercept shifters (Δh_i) is zero.

The Pigouvian taxes (τ_i) introduced in *Scenario 1* and *Scenario 3* are calculated as:

$$\tau_i = CO_2e \; x \; P_{CO_2e} \tag{10}$$

That is, the product between the commodity groups' average climate footprint (CO_2e) and the Swedish carbon tax (P_{CO_2e}), which currently is set to 1.2 SEK/kg CO_2e (Government Offices of Sweden, 2022). In *Scenario 2*, the *Eco-labelled* commodity group is assigned a subsidy for the purpose of achieving a substantial increase in total seafood consumption from sustainable practices. By constructing a linear demand curve system, we utilize that price interventions are not uniformly (albeit linearly) distortionary. This allows for calibrating subsidies, discerning its expected impact on demand and GHG emissions, and settle on a subsidy level that is aligned with the intervention's objective. Therefore, we assess the impact of exempting this commodity group from the VAT. Such an intervention is analogous to a 12 percent subsidy. Considering the large share of imported seafood, there is a risk of carbon leakage abroad from the domestic governmental interventions. In a deliberate precautionary measure, as suggested by Wirsenius et al. (2010), we levy corrective taxes on consumers, rather than producers.

Conducting the analysis using producer-specific emissions and interventions, may incentivize individualistic decarbonization actions and improve the cost-effectiveness of the emission reductions. However, this would impose significant sampling- and administration costs. Instead, we rely on average emissions for products reaching Swedish retailers' gate and implicitly neglects variation in emission intensity across producers. Similar to Wirsenius et al. (2010), we argue that using average emissions enhance the administrative cost-effectiveness of the GHG reductions from food products, and the introduced bias is minor (ibid).

After the policy has altered the price of the jth good (Δp_j), we simulate intercept shifts of substitute and complementary goods (Δh_i) as shown in the following equation:

$$\Delta h_i = \sum \Delta p_j \frac{\varepsilon_{ij}^{M*} q_i^0}{p_j^0} + \sum \Delta p_r \frac{\varepsilon_{ru}^{M} q_u^0}{p_r^0} s_u \tag{11}$$

Equation (11) is readily applicable to the third level in the demand system, whereas third level terms drop out as we transition upwards to the second level in the demand system. Subsequently, the policy-induced change in consumption ($\Delta q = q_i^1 - q_i^0$) multiplied with the commodity group's climate footprint yields each policy scenario's impact on the sector's GHG emissions.

5. Results

Subsections below summarizes this study's results based on the methodology outlined in Chapter 4. First, price- and income elasticities are presented in the first subsection. Uncompensated elasticities are calculated in TSP, using equation (2) through (6). Final compensated elasticities are obtained by applying equation (7) and (8) to the TSP output in Excel. Second, three policy scenarios are analyzed using equation (9) through (11). Emphasis is placed on assessing the feasibility of addressing the market failures without conflicting with official health recommendations. Policy-interventions' impacts on demand and GHG emissions primarily refer to daily impacts at the store. Lastly, the results' sensitivity to selected climate footprints is tested in the terminal subsection.

5.1 Price- and Income Elasticities of Demand

This subsection hinges upon the two-stage budgeting process and methodology outlined in section 4.2 and 4.3. The own-price elasticity for seafood at the first level amounts to -0.491 and is obtained from Säll et al. (2020). In contrast, second and third level estimates are calculated by the author of this study.

First, we specified four EQUAIDS functions for the regressions at the second as well as the third level. That is, one function for each commodity group. Since autocorrelation issues arose in initial EQUAIDS specifications, we exercised two interventions in the functional form. First, the time trend variable had to be omitted. Similar to Alessie & Kapteyn (1991), we argue that this specification is adequate if one is willing to assume that consumers are time inconsistent in their habit formation. Accordingly, utility functions shift myopically and a time trend variable is thus deemed insufficient. The relative short time period analyzed in this study further adds to this decision. Second, the expenditure-, aggregated price index and certain commodity price variables had to be lagged one or two days on each level in the demand system. Cortinhas & Black (2014, p.657) explain that lagging independent variables is a common solution to overcome autocorrelation issues and improves coherence with previous consumption patterns (i.e. autoregressive modeling). The final specification of the EQUAIDS functions yielded 18 out of 30 estimates statistically significant at the 1 percent level. Each model obtained R^2 between 0.05 and 0.44, signaling a low to moderate statistical fit. Lastly, the Lagrange Multiplier (LM) heterogeneity tests show that the initial issues with autocorrelation were successfully alleviated to tolerable levels[4]. A full disclosure of test results is found in *Table A3* and *Table A4,* in the Appendix.

Table 4 shows the compensated-, followed by the final uncompensated elasticities estimated in this study. Concerning compensated elasticities, all own-price elasticities on the second level exhibit negative values as expected. *Fish, Canned Seafood* and *Breaded Seafood* display elastic demand with own-price elasticities slightly lower than -1. The income elasticity for *Fish* is slightly higher than 1, meaning it may be perceived as a luxury good, which is reasonable given its relatively high price per unit. In contrast, consumers consider *Canned Seafood* and *Breaded Seafood* as necessities. This is sensible considering their relatively low prices per kilogram and our perception of the goods being viewed as lower quality by consumers. Additionally, *Shellfish* exhibit inelastic demand whilst the income elasticity suggests it is a luxury good. This seems reasonable since the shellfish group primarily consists of crustaceans and are sold to the highest mean price at the second level. Lastly, the cross-price elasticities for Fish suggests that *Canned Seafood* and *Breaded Seafood* are substitute goods whereas *Shellfish* is a complementary good.

Compensated elasticities on the third level shows that *Eco-labelled* fish is the only luxury good, and the product with the strongest price sensitivity. *Bottom trawled* and *Conventionally produced* fish show elastic demand, and income elasticities suggesting they are necessities. *Ecotoxic* fish is estimated to have an income elasticity and an own-price elasticity suggesting it is a necessity with inelastic demand. Since this product group includes relatively expensive and sought-after fish, we expected the income elasticity and own-price elasticity to be higher. It is considered likely that its low quantities consumed could be a signal of these species primarily being purchased at e.g. fishmongers. If proven true, the elasticities in this study would not retain the true elasticities for ecotoxic fish. Cross-price elasticities suggest *Eco-labelled*, *Conventionally produced* and *Bottom trawled* fish are substitutes. This is expected since the former two product groups comprise similar species and *Bottom trawled* fish consists of eco-labelled and conventionally produced fish. Furthermore, the cross-price elasticities between *Canned*- and *Breaded Seafood* suggest they are consumed as complements. Although this might seem counterintuitive at first glance, we argue it is not. Complementarity could arise from the desire to have a diversified diet. Lastly, it is found that bottom trawled and ecotoxic fish are considered complements. A potential explanation is that this result is driven by a preference for consuming species found in Swedish waters. In particular, Perch, Zander, and Baltic herring (i.e. *Ecotoxic* products) as well as Plaice and Cod (i.e. *Bottom trawled),* can all be caught in Swedish waters.

[4] It may be worth noting, however, that the Durbin-Watson test statistics indicate autocorrelation was not terminated completely.

Final uncompensated own-price elasticities on the second level are somewhat smaller and cross-price elasticities somewhat larger than the compensated elasticities since they acknowledge the multi-stage budgeting process. Consequently, the own-price elasticity of *Fish* now suggests its demand is inelastic whereas the rationale for the other second stage own-price elasticities remain. Regarding final uncompensated income elasticities, the estimate for *Fish* is slightly lower than 1 meaning it is perceived as a necessity. The rationale for the other commodity groups on this level is unaltered. In addition, after acknowledging the two-stage budgeting process the cross-price elasticity between *Breaded Seafood* and *Fish* is slightly larger than the own-price elasticity of *Breaded Seafood*. Although this is surprising, the difference between the two elasticities occurs on the first decimal and replacing the cross-price elasticity to a smaller value proved to have a negligible impact on the results.

The third level's final uncompensated elasticities show the same sign and similar magnitudes as for the compensated elasticities. Thus, the interpretation of the compensated elasticities carries over to the final elasticities throughout.

Table 4. Compensated and final uncompensated elasticities of demand

Compensated Elasticities					
SECOND LEVEL	Fish	Canned Seafood	Breaded Seafood	Shellfish	Income
Fish	-1.022*** (0.047)	0.075*** (0.016)	0.077*** (0.015)	-0.149** (0.059)	1.019*** (.039)
Canned Seafood	0.728*** (0.114)	-1.153*** (0.104)	-0.262*** (0.071)	0.004 (0.108)	0.683*** (0.072)
Breaded Seafood	0.863*** (0.121)	-0.294*** (0.083)	-1.087*** (0.083)	-0.092 (0.129)	0.610*** (0.086)
Shellfish	-0.415*** (0.117)	-0.038 (0.031)	-0.065** (0.032)	-0.646*** (0.174)	1.164*** (0.114)
THIRD LEVEL	Eco-labelled	Bottom trawled	Conventionally produced	Ecotoxic	Income
Eco-labelled	-2.114*** (0.125)	0.135 (0.085)	0.806 (0.099)	-0.049*** (0.038)	1.221*** (0.084)
Bottom trawled	0.267** (0.119)	-1.141*** (0.156)	0.280*** (0.100)	-0.161*** (0.056)	0.754*** (0.077)
Conventionally produced	0.243*** (0.028)	0.022 (0.021)	-1.286*** (0.029)	0.023*** (0.009)	0.998*** (0.023)
Ecotoxic	-0.143 (0.163)	-0.496*** (0.172)	0.479*** (0.135)	-0.696*** (0.112)	0.856*** (0.110)
Final Uncompensated Elasticities					
SECOND LEVEL	Fish	Canned Seafood	Breaded Seafood	Shellfish	Income
Fish	-0.720	-0.134	-0.142	-0.173	0.947
Canned Seafood	1.098	-1.124	-0.240	0.155	0.634
Breaded Seafood	1.197	-0.265	-1.065	0.046	0.567
Shellfish	0.115	-0.092	-0.131	-0.490	1.082
THIRD LEVEL	Eco-labelled	Bottom trawled	Conventionally produced	Ecotoxic	Income
Eco-labelled	-2.055	0.177	1.035	-0.035	1.156
Bottom trawled	0.303	-1.115	0.421	-0.152	0.715
Conventionally produced	0.291	0.056	-1.099	0.035	0.945
Ecotoxic	-0.102	-0.467	0.639	-0.686	0.810

Source: Own calculations using data from ICA Maxi Nacka.
Notes: Rows display the ith [rth] goods and columns the jth [uth] goods, with standard errors reported in parentheses. Statistical significance: *** p<0.01, ** p<0.05, and * p<0.1

5.2 Policy Implications on Demand and GHG Emissions

Table 5 shows the impact of the emission-tax on *Bottom trawled* fish, as considered in *Scenario 1*. Equation (10) suggests that the adequate emission-adapted tax on bottom trawled fish should be set to 9.45 SEK/kg (i.e. a 5.48 percent price increase). Subsequent impacts on consumption and kilograms of CO_2e (i.e. GHG) emissions are calculated using Equation (9) and (11) and refer to average daily changes. After the introduction of the emission-tax, it is estimated that consumption of *Bottom trawled* fish would decrease by 6.11 percent and GHG emissions by 20.29 kg. Consumption of *Eco-labelled* and *Conventionally produced* fish are expected to increase by 0.97 and 0.31 percent, respectively. Consequently, GHG emissions would rise by 1.93 and 2.31 kg for the two commodity groups, respectively. Contrarily, the consumption of *Ecotoxic* products is expected to decrease by 2.56 percent which would reduce GHG emissions by 1.21 kg. Altogether, these changes are estimated to reduce consumption of *Fish* by 0.42 percent and associated GHG emissions by 17.26 kg.

Second level impacts indicate a shift into all other commodity groups and a modest dampening effect on the reduction in GHG emissions achieved on the third level. The strongest impact is found for consumption of *Breaded*- and *Canned Seafood* which are expected to increase by 0.05 percent, respectively. Substitution into *Shellfish* amounts to a 0.02 percent increase in consumption. Considering these substitutions, the reduction in consumption of *Seafood* amounts to 0.23 percent whilst the reduction in GHG emissions is down to 16.76 kg.

Table 5. Scenario 1 policy implications on demand and GHG emissions

THIRD LEVEL	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Eco-labelled	0	0	63.93	64.55	0.97	198.40	200.33	1.93	0.97
Bottom trawled	9.45	5.48	42.15	39.57	-6.11	331.92	311.63	-20.29	-6.11
Conventionally produced	0	0	255.33	256.11	0.31	756.15	758.46	2.31	0.31
Ecotoxic	0	0	15.59	15.19	-2.56	47.26	46.05	-1.21	-2.56
TOTAL CHANGE	1.16	0.56	377.00	375.43	-0.42	1333.73	1316.47	-17.26	-1.31
SECOND LEVEL	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Fish	1.16	0.56	377.00	375.43	-0.42	1333.73	1316.47	-17.26	-1.31
Canned Seafood	0	0	49.26	49.28	0.05	241.37	241.49	0.12	0.05
Breaded Seafood	0	0	42.05	42.07	0.05	226.55	226.65	0.11	0.05
Shellfish	0	0	182.09	182.12	0.02	1581.28	1581.54	0.26	0.02
TOTAL CHANGE	0.68	0.35	650.40	648.90	-0.23	3382.92	3366.15	-16.76	-0.50

Source: Own calculations using data from ICA Maxi Nacka and climate footprints from Carbon Cloud (2022).
Notes: All numerical values express the estimated impact of the emission-tax on a representative day. Quantities are expressed in kilograms, prices in SEK per kg, and GHG emissions in kilograms of CO_2e.

Table 6 presents the impacts on demand and GHG emissions from adding a subsidy on *Eco-labelled* products to the policy design. The magnitude of our proposed subsidy is determined by first finding the level at which the Seafood consumption commence increasing. Our analysis shows that any subsidies larger than 5.8 SEK/kg (i.e. 3.4 percent price reduction) counteracts the tax-induced demand reduction of seafood entirely. For the purpose of achieving a substantial increase in seafood consumption we assess the impact of a subsidy at 20.33 SEK/kg (i.e. 12 percent price reduction) in *Scenario 2*. Thus, the tax-subsidy scheme applies the same tax level on Bottom trawled fish whilst subsidizing eco-labelled fish by 12 percent. Following this, *Bottom trawled* fish consumption is estimated to decrease by 9.75 percent and GHG emissions by 32.35 kg. Furthermore, consumption of *Eco-labelled* fish is estimated to increase by 25.64 percent and GHG emissions by 50.86 kg. In addition, consumption of *Conventionally produced* and *Ecotoxic* fish is estimated to decrease by 3.18 and 1.34 percent, respectively. Associated reductions in GHG emissions amount to 24.08 and 0.63 kg, respectively. In total, the substitution within the *Fish* product group increases total consumption by 1.05 percent whilst achieving a reduction in GHG emissions by 6.20 kg. This reduction is primarily attributable to a reinforced reduction in consumption of bottom trawled fish, triggered by the subsidy on *Eco-labelled* fish.

Second-level substitution shows that consumption of all other product groups decreases after the tax-subsidy scheme is introduced. Despite lower consumption, the magnitude of the impact is modest. *Canned Seafood* exhibits the largest relative reduction in consumption at 0.10 percent, followed by *Breaded Seafood* at 0.09 percent. Consequential reductions of GHG emissions amounts to 0.24 and 0.21, respectively. Consumption of *Shellfish* is estimated to decrease by 0.03 percent, and associated GHG emissions by 0.52 kg. Overall, these results suggests that *Seafood* consumption would increase by 0.58 percent whilst GHG emissions would decrease by 7.18 kg.

Table 6. Scenario 2 policy implications on demand and GHG emissions

THIRD LEVEL	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Eco-labelled	-20.33	-12.00	63.93	80.32	25.64	198.40	249.26	50.86	25.64
Bottom trawled	9.45	5.48	42.15	38.04	-9.75	331.92	299.57	-32.35	-9.75
Conventionally produced	0	0	255.33	247.20	-3.18	756.15	732.07	-24.08	-3.18
Ecotoxic	0	0	15.59	15.38	-1.34	47.26	46.63	-0.63	-1.34
TOTAL CHANGE	-2.28	-1.10	377.00	380.94	1.05	1333.73	1327.53	-6.20	-0.47
SECOND LEVEL	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Fish	-2.28	-1.10	377.00	380.94	1.05	1333.73	1327.53	-6.20	-0.47
Canned Seafood	0	0	49.26	49.21	-0.10	241.37	241.13	-0.24	-0.10
Breaded Seafood	0	0	42.05	42.01	-0.09	226.55	226.33	-0.21	-0.09
Shellfish	0	0	182.09	182.03	-0.03	1581.28	1580.75	-0.52	-0.03
TOTAL CHANGE	-1.33	-0.68	650.40	654.19	0.58	3382.92	3375.74	-7.18	-0.21

Source: Own calculations using data from ICA Maxi Nacka and climate footprints from Carbon Cloud (2022). *Notes:* All numerical values express the estimated impact of the emission-tax on a representative day. Quantities are expressed in kilograms, prices in SEK per kg, and GHG emissions in kilograms of CO_2e.

In *Scenario 3*, we introduce emission-taxes to all products except *Eco-labelled* in an attempt to achieve the largest possible GHG reductions from seafood consumption. Since our simulation showed that a VAT exemption on *Eco-labelled* products would have a negligible effect on the reduction of overall seafood consumption, we chose to exclude a subsidy in our design of this policy scenario. Another essential point is that our simulation suggests that taxing all third-level commodity groups conflict with two of three dietary recommendations. *Table 7* shows that the introduction of emission-taxes is estimated to decrease consumption of taxed products and increase consumption of Eco-labelled fish. The emission-tax on *Bottom trawled* fish is unaltered and is estimated to cause a 5.65 percent reduction in consumption. This is equivalent to a decrease of associated GHG emissions by 18.74 kg. After introducing the emission-tax of 3.55 SEK/kg (i.e. a 1.57 percent price increase) on *Conventionally produced* fish, it is estimated that consumption would decrease by 1.37 percent. The associated reduction of GHG emissions amounts to 10.39 kg. The emission-tax on *Ecotoxic* fish amounts to a 3.64 SEK/kg (i.e. 1.28 percent) price increase and is estimated to decrease consumption by 2.43 percent. This is equivalent to a decrease of associated GHG emissions by 1.15 kg. In contrast, the tax-scheme is estimated to achieve a consumption increase of *Eco-labelled* fish at 2.55 percent. Consequently, associated GHG emissions is estimated to increase by 5.06 kg. In total, the substitution across products inside the *Fish* group is estimated to reduce consumption by 1.23 percent and GHG emissions by 25.23 kg.

Second-level substitution is larger, relative to policy Scenario 1. The analysis shows that consumption of *Canned-* and *Breaded Seafood* is estimated to increase by 0.16 and 0.15 percent and associated GHG emissions would increase by 0.39 and 0.35 kg, respectively. Consumption of *Shellfish* is estimated to increase by 0.05 percent and associated GHG emissions by 0.84 kg. Altogether, the simulation suggests *Seafood* consumption would decrease by 0.68 percent and associated GHG emissions by 23.65 kg.

Table 7. Scenario 3 policy implications on demand and GHG emissions

THIRD LEVEL	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Eco-labelled	0	0	63.93	65.56	2.55	198.40	203.46	5.06	2.55
Bottom trawled	9.45	5.48	42.15	39.77	-5.65	331.92	313.18	-18.74	-5.65
Conventionally produced	3.55	1.57	255.33	251.82	-1.37	756.15	745.75	-10.39	-1.37
Ecotoxic	3.64	1.28	15.59	15.21	-2.43	47.26	46.11	-1.15	-2.43
TOTAL CHANGE	3.68	1.77	377.00	372.36	-1.23	1333.73	1308.50	-25.23	-1.93
SECOND LEVEL	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Fish	3.68	1.77	377.00	372.36	-1.23	1333.73	1308.50	-25.23	-1.93
Canned Seafood	0	0	49.26	49.34	0.16	241.37	241.75	0.39	0.16
Breaded Seafood	0	0	42.05	42.12	0.15	226.55	226.89	0.35	0.15
Shellfish	0	0	182.09	182.19	0.05	1581.28	1582.12	0.84	0.05
TOTAL CHANGE	2.15	1.10	650.40	646.00	-0.68	3382.92	3359.26	-23.65	-0.70

Source: Own calculations using data from ICA Maxi Nacka and climate footprints from Carbon Cloud (2022).
Notes: All numerical values express the estimated impact of the emission-tax on a representative day. Quantities are expressed in kilograms, prices in SEK per kg, and GHG emissions in kilograms of CO_2e.

Regarding the obtained result, it is evident that *Scenario 1* and *Scenario 3* improve compliance with two dietary recommendations by achieving increased consumption of *Eco-labelled,* and reduced demand of *Bottom trawled* and *Ecotoxic* fish. Despite achieving an increase (decrease) of environmentally friendly (inferior) fish, the policy scenarios contradict the first dietary recommendation of increased seafood consumption. Another potentially undesired impact from these policy scenarios are the increases of *Shellfish*, *Canned-*, and *Breaded Seafood* since all commodity groups on the second level in the demand system contain bottom trawled seafood. Fortunately, substitutions into these commodity groups are modest, at most. Additionally, *Scenario 3* showed that taxing all third-level commodity groups except *Eco-labelled* fish is superior in terms of GHG reductions.

Furthermore, *Scenario 2* has the potential to reinforce the lower demand of bottom trawled fish whilst increasing overall seafood consumption. However, the introduction of the subsidy dampens the reduction on consumption of *Ecotoxic* fish. Nonetheless, the tax-subsidy policy outperforms the tax policies in terms of compliance with dietary recommendations at the expense of a somewhat lower reduction of GHG emissions.

Additionally, we conducted a back-of-the-envelope calculation for the policy scenarios' implication on daily nationwide emissions. By dividing the daily averages for seafood sales at the store (i.e. 646 996 grams) by consumption per person (i.e. 34.29 grams; Borthwick et al. 2019), it is calculated that 18 870 seafood consumers visit the store. Next, we use total emission reductions at the store and Sweden's population at 10.45 million (Statistics Sweden, 2021) to develop a ballpark figure for nationwide GHG reductions. As a result, daily nationwide GHG reductions from *Scenario 1*, *Scenario 2*, and *Scenario 3* would be in the ballpark around 9 280kg, 3 980 kg, and 13 100 kg, respectively.

5.3 Sensitivity Analysis

In the sensitivity analysis we test the sensitivity of the estimated policy impacts by including more species in our calculation of weighted climate footprints. Moberg et al. (2019) provide the necessary estimates in their supplementary material. However, leakage from the ozone depleting cooling medium R22 were omitted since this cooling medium has been phased out since 2015 (the Swedish Environmental Protection Agency, 2022)[5]. Merging climate footprints from the two sources is considered feasible since both sources use GWP_{100} and ALCA and may improve the precision of the governmental intervention for two reasons. First, the extension adds climate footprints for Saithe, Alaska pollock, Rainbow trout, Saithe,

[5] According to the Swedish Transport Agency, data on which cooling medium fishermen have switched to is not yet published (personal communication, STA, January 27th , 2022).

and Pangasius. Second, it complements the climate footprints for Northern prawn, Cod, Herring, Mackerel, and Plaice. The greatest source of uncertainty is the climate footprint of *Bottom trawled* Plaice which is estimated to be 11 kg CO_2e in Carbon Cloud (2022) and 22.07 kg CO_2e in Moberg et al. (2019). In general, climate footprints are pervasively larger in the latter source. Nonetheless, merging climate footprints from two sources requires making the additional assumption that both sources utilize identical procedures in their calculations.

Figure 4 shows the relative differences of this study's policy impacts under the two approaches for climate footprints[6]. Overall, deviations between the two scenarios are minor in terms of percentage point (ppt) and occasionally major in terms of percent. Concerning *Scenario 1,* the merged climate footprints suggest the emission-tax on *Bottom trawled* fish should be set somewhat higher at 10.27 SEK/kg. Consequently, impacts on demand for commodity groups deviate between 0.09 and -0.53 ppt, as opposed to the baseline scenario. Under the merged climate footprints, the magnitude and direction for third-level commodity groups are reinforced by circa 9 percent, respectively. As a result, *Fish* consumption is further reduced under the merged climate footprints. Second-level substitution is unaltered, and the estimated demand decrease of *Seafood* is 9 percent larger, as opposed to the baseline scenario. Consequential impacts on GHG emissions from commodity groups deviate between 1.41 kg and -3,66 kg, relative to the baseline scenario. The largest deviations are found for *Ecotoxic* and *Eco-labelled* fish who exhibit emission impacts circa 58 percent larger than in the baseline scenario. Altogether, GHG emissions from *Seafood* consumption is 10 percent lower, relative to the baseline scenario.

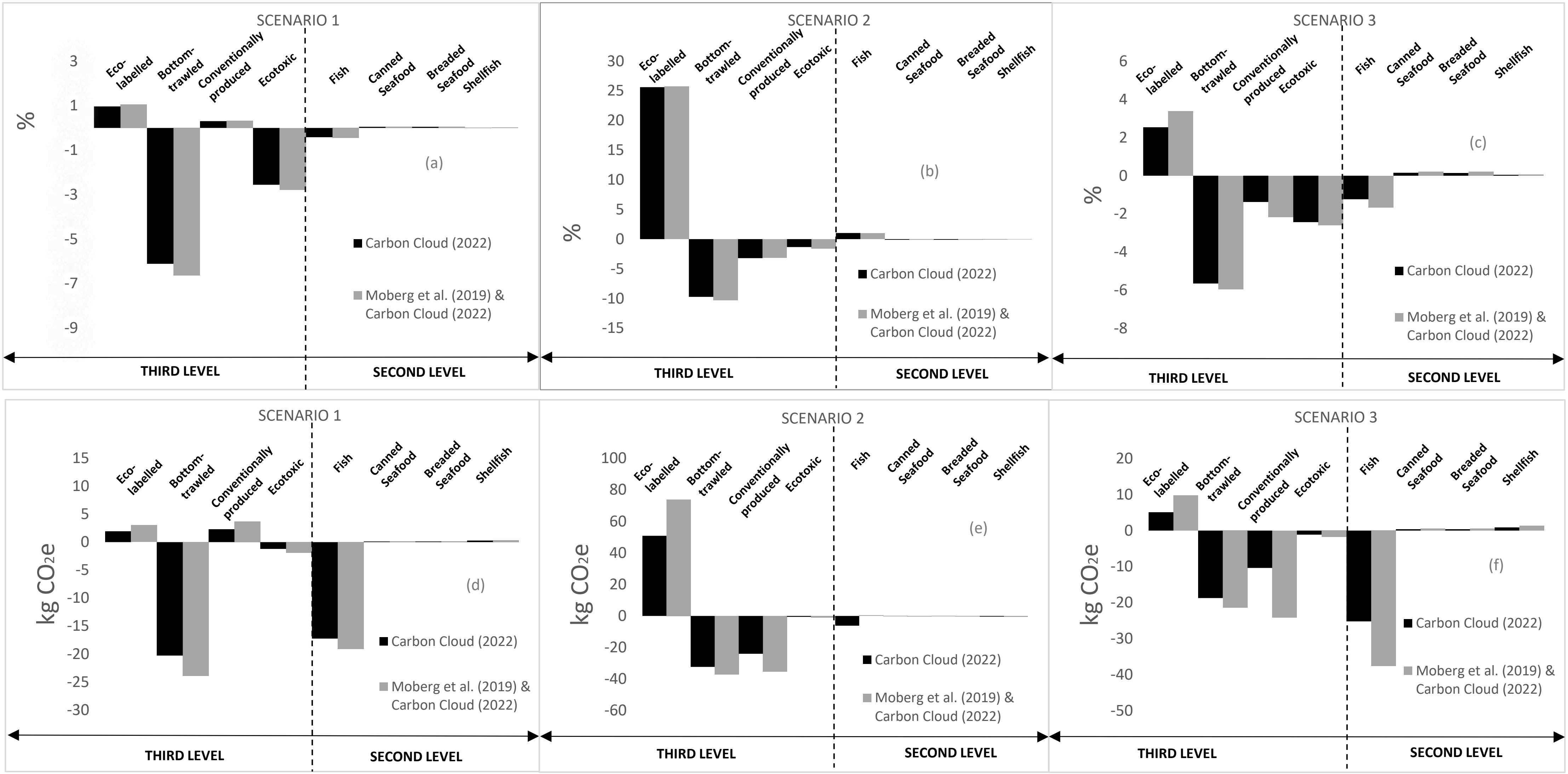
The subsidy on *Eco-labelled* fish added in *Scenario 2* is unaltered by the merged climate footprints. Impacts on demand for commodity groups deviate between 0.08 and -0.53 ppt. Under the merged climate footprints, demand of *Bottom trawled* and *Ecotoxic* fish are reduced another 5 and 16 percent, respectively. Altogether, *Fish* consumption increases somewhat less, and second-level substitution is negligibly affected. In turn, *Seafood* consumption decreases by 2 percent, relative to the baseline scenario. Consequential impacts on GHG emissions from commodity groups deviate between 22.99 kg and -11.27 kg, relative to the baseline scenario. The magnitude and direction of GHG emissions are reinforced by 71 percent for *Ecotoxic* fish and 45 percent for *Eco-labelled* fish, relative to the baseline scenario. Third-level substitution yields a slight increase in GHG emissions, whereas a decrease is achieved in the baseline scenario. However, second-level substitution results in reduced GHG emissions from *Seafood* consumption, albeit at a 90 percent lower level.

[6] All numerical values from the sensitivity analysis can be found in Table A5.

Emission-taxes in *Scenario 3* are pervasively larger under the merged climate footprints, and are set to 10.27, 5.26, and 5.33 SEK/kg, for *Bottom trawled*, *Conventionally produced*, and *Ecotoxic* fish, respectively. As a result, impacts on demand for commodity groups deviate 0.84 to -0.79 ppt. In particular, demand of *Conventionally produced* fish decreases yet another 58 percent and *Eco-labelled* fish increase 33 percent more, as opposed to the baseline scenario. Third-level impacts result in a 35 percent larger reduction of *Fish* consumption. Second-level substitution is still modest and overall *Seafood* consumption decreases 34 percent, relative to the baseline scenario. Reductions of GHG emission deviate 4.68 to -13.75 kg, as opposed to the baseline scenario. Emission reductions from *Conventionally produced* and *Ecotoxic* fish are 132 percent and 56 percent larger, respectively. Contrarily, GHG emissions from *Eco-labelled* fish increase another 92 percent. After somewhat enlarged substitution into second-level products, emissions from *Seafood* consumption are reduced another 49 percent.

The back-of-the-envelope calculation for daily nationwide GHG reductions from *Scenario 1*, *Scenario 2*, and *Scenario 3* amounts to 10 253 kg, 388 kg, and 19 508 kg, respectively.

Figure 4. Sensitivity analysis for climate footprints.



Source: Own illustration using data from ICA Maxi and estimates from Carbon Cloud (2022) and Moberg et al. (2019).
Notes: Figure (a) through (c) shows percentage change in demand and Figure (d) through (f) shows associated impact on GHG emissions after the introduction of each policy scenario. Illustrations exhibit the impacts on a representative day.

6. Discussion

This study's findings both modify and adds to the existing literature on seafood demand. The modification is attributable to the novel policy scenarios introduced for differentiated fish products, whereas the elasticities add to the existing literature. Despite this limitation for a scientific comparison to the policy literature, it is feasible to perceptively discuss this study's elasticities, in relation to previous scientific literature.

In general, our estimated own-price elasticities are elastic. These results are comparable to other revealed preference studies on seafood demand (see Bronnman, 2016; Bronnman et al., 2019; Singh et al., 2012; Surathkal et al., 2017). Craig (2009) conducted an extensive meta-analysis of own-price elasticities obtained from different seafood demand system approaches. The author presents median own-price elasticities and identify study-specific drivers of the magnitude of own-price elasticities. It is found that QUAIDS studies have somewhat higher elasticities, as opposed to linear AIDS estimations. Contrarily, panel data typically yields somewhat lower own-price elasticities for fish, relative to when cross-sectional data is being used. In general, the median own-price elasticities for fish and shellfish amounts to -0.79 and -0.86, respectively. Thus, our final uncompensated own-price elasticity for *Fish* at -0.72 suggests that the bidirectional forces induced by our approach does not yield an anomalistic result. However, our estimated final uncompensated own-price elasticity for *Shellfish* at -0.49 is substantially lower than the median estimate in the pool of research (Craig, 2009). Although it may be a consequence of stable consumption of shellfish (primarily Northern shrimp) in Sweden, it is not firmly established why our own-price elasticity for shellfish is smaller than in previous studies. This deviation is an important result since it supports the school of though claiming seafood demand patterns can vary substantially across countries and regions (see Bronnman, 2016; Craig., 2009; Johnston et al., 2006).

Our cross-price elasticities between *Breaded Seafood* and *Fish* suggests that the two commodity groups are consumed as substitutes. Thus, if the price of breaded seafood increases, the cross-price elasticity with fish at 0.86 suggests consumers would switch to consuming more fish instead. This contradicts the findings in Bronnman et al. (2019) who find that fresh and frozen fish is a complementary good to breaded seafood. This divergence could stem from the authors' delimitation to Germany and stated preference data. In contrast, this study is delimited to Sweden and relies on revealed preference data.

7. Conclusion

The strongest finding of this study is that green tax- and subsidy interventions does not necessarily jeopardize dietary recommendations' fulfilment. However, solely distorting consumption on fish is insufficient for achieving the recommended consumption increase of seafood at 25 percent. Despite achieving desirable substitution towards fish products with relatively low emission-intensity, the second-level substitution is modest. Therefore, we stress that analogue policy-analyses for substitute goods on the second level in our demand system are needed to come closer to achieving public dietary recommendations.

Scenario 1 introduces an emission-tax at 9.45 SEK/kg (i.e. a 5.48 percent price increase). Our simulations suggests that the policy intervention's impact on demand of *Bottom trawled* and *Ecotoxic* fish amount to -6.11 and -2.56 percent, respectively. Associated impact on GHG emissions amount to -20.29 and -1.21 kg. Contrarily, *Eco-labelled* fish consumption is estimated to increase by 0.97 percent. This would increase associated GHG emissions by 1.93 kg. The estimated impact of the considered emission-tax on *Seafood* consumption amounts to -0.23 percent. Consequential impact on GHG emissions amounts to -16.76 kg per day. Despite achieving increased compliance with two dietary recommendations, the policy scenario contradicts the first dietary recommendation of increased seafood consumption.

Scenario 2 introduces a 12 percent subsidy on *Eco-labelled* products, in addition to the unaltered emission-tax on bottom trawled fish. It is estimated that the tax-subsidy scheme has the potential to affect consumption and GHG emissions from *Bottom trawled* fish by -9.75 percent and -32.35 kg, respectively. Furthermore, consumption and associated GHG emissions from *Eco-labelled* fish are expected to increase by 25.64 percent and 50.86 kg, respectively. Additionally, the introduction of the subsidy dampens the reduction on consumption and GHG emission from *Ecotoxic* fish to -1.34 percent and -0.63 kg. The net effect of the third- and second level substitution yields a 0.58 increase of *Seafood* consumption. Despite the increase in consumption, GHG emissions change by -7.18 kg. Thus, the tax-subsidy policy increases compliance with the two last recommendations without conflicting with the first.

Scenario 3 introduces emission-taxes for all products except for the *Eco-labelled* commodity group. Our simulation indicates that this tax scheme may be capable of achieving an increase of *Eco-labelled* products at 2.55 percent whilst reducing consumption of *Bottom trawled* and *Ecotoxic* fish by -5.65 and -2.43 percent, respectively. Corresponding impacts on GHG emissions amount to 5.06 kg, -

18.74kg, and -1.15kg, respectively. Subsequent substitution is estimated to affect *Seafood* consumption by -0.68 percent. Associated impact on GHG emissions is estimated to amount to -23.65 kg. Thus, this policy intervention is estimated to increase compliance with the second and third dietary recommendations, at the expense of the first recommendation (i.e. overall increase in seafood consumption).

Nationwide daily GHG reductions from altered seafood consumption from *Scenario 1, Scenario 2,* and *Scenario 3* are in the ballpark around 9 280kg, 3 980 kg, and 13 100 kg, respectively. The merged climate footprints approach suggests somewhat higher reductions for the first and third scenarios, and lower reductions for the second scenario. Nonetheless, these are gross calculations, and a more formal calculation is encouraged.

One important condition for our results is the magnitude of climate footprints. It is found that a slight increase of climate footprints has a more modest impact on demand than on the achieved GHG reductions. An important premise for our result is the disaggregation into commodity-groups on the third level. Yet, the stated preference literature on Swedish seafood consumption is insufficient for testing its validity. Another caveat to our results is that bottom trawled products were found in all second-level commodity groups, meaning that second-level substitution pose a risk of countering our intentions. Similarly, first-level substitution could have an impact on this study's results. These aspects are beyond the scope of this study, but we encourage future studies to explore this. Moreover, the interventions' distributional effects and potentially regressive impacts would be an appealing extension to our study. Although each policy scenario positively influence two externalities, CO_2e emissions and overfishing, they do not consider externalities such as high-grading, congestion, and bycatches. Hence, they attain second-best optimums.

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Appendix

Table A1. Aggregation into product groups

Product group	Species
Fish	Salmon, Atlantic cod, Arctic char, Ling, Plaice, Tuna, Baltic herring, Pangasius, Haddock, Seabream, Zander, Halibut, Mackerel, Sockeye salmon, Saithe, Wolffish, Turbot, Alaska pollock, Rainbow trout, Perch, Sea bass, Witch, Monkfish, Whitefish, Atlantic pollock, Rose fish, Lemon sole, Atlantic herring, Clarias, Trout, Nile tilapia, Cusk, Hake, Sole, Greenland Halibut, Yellowtail,
Canned Seafood	Tuna, Mackerel, Sardine, Chilean mussel, Baltic herring, Anchovy, Blue mussel, European sprat, Blue crab, Venus clams, and Horse mackerel.
Breaded Seafood	Atlantic cod, Salmon, Squid, and Plaice
Shellfish	Northern shrimp, Blue mussel, Lobster, Turkish crayfish, Signal crayfish, Langoustine, Whiteleg shrimp, Brown crab, Red swamp crayfish, Common cockle, Scallop, King crab, Squid, Great scallop, Venus clam, Argentine red shrimp, Green-lipped mussel, Razor shell, King shrimp, and Pasiphaea.
Eco-labelled	Salmon, Atlantic cod, Ling, Pangasius, Sockeye salmon, Alaska pollock, Rainbow trout, Saithe, Haddock, and Atlantic herring.
	Salmon, Alaska pollock, Pangasius, Haddock, Ling, Rainbow trout, Atlantic pollock, Atlantic herring, Sockeye salmon, and Atlantic cod.
Bottom trawled	Atlantic cod and Plaice
Conventionally produced	Salmon, Atlantic cod, Arctic char, Haddock, Seabream, Mackerel, Wolffish, Turbot, Alaska pollock, Rainbow trout, Sea bass, Witch, Monkfish, Whitefish, Rose fish, Lemon sole, Atlantic herring, Clarias, Trout, Nile tilapia, Cusk, Hake, Sole, Greenland Halibut, Yellowtail,
Ecotoxic	Tuna, Baltic herring, Zander, Halibut, and Perch.

Notes: All species are listed from most to least sold.

Table A2. Species included in calculations for climate footprint averages

	Carbon Cloud (2022)	Moberg et al. (2019) & Carbon Cloud (2022)
Average: Seafood		
Average: Fish		
Average: Canned Seafood	Tuna	Tuna
Average: Breaded Seafood	Atlantic cod, Plaice, and Salmon	Atlantic cod, Plaice, and Salmon
Average: Shellfish	Northern shrimp and Langoustine	Northern shrimp and Langoustine
Average: Eco-labelled	Atlantic cod, Salmon, and Atlantic herring.	Atlantic cod, Salmon, Atlantic herring, Saithe, Alaska pollock, Rainbow trout, Saithe, and Pangasius
Average: Bottom trawled	Atlantic cod and Plaice	Atlantic cod and Plaice
Average: Conventionally produced	Salmon, Atlantic cod, Atlantic herring, and Mackerel	Salmon, Atlantic cod, Atlantic herring, Mackerel, Saithe, and Alaska pollock.
Average: Ecotoxic		

Notes: The climate footprints for Fish and Seafood are intentionally left blank since they are implicitly determined by lower-level aggregation.

Table A3. Test statistics for the demand system outlined in this study

	SECOND LEVEL			**THIRD LEVEL**		
	EQUAIDS 1 Dep. var: s_1	EQUAIDS 2 Dep. var: s_2	EQUAIDS 3 Dep var: s_3	EQUAIDS 1 Dep. var: s_1	EQUAIDS 2 Dep. var: s_2	EQUAIDS 3 Dep var: s_3
Mean (dep. var)	.584	.082	.072	.169	.123	.667
Std. dev. (dep. var)	.103	.027	.029	.074	.041	.080
SSR	2.376	.162	.177	.924	.398	1.074
Variance of residuals	.982E-02	.669E-03	.731E-03	.382E-02	.165E-02	.444E-02
Std. error of regression	.099	.026	.027	.062	.041	.067
R^2	.084	.106	.117	.297	.045	.300
LM-heterogeneity test	1.536 [.215]	1.593 [.207]	.024 [.877]	1.592 [.207]	.122E-02 [.972]	2.630 [.105]
Durbin Watson	.804	1.120	1.016	1.008	.790	.930

Source: Own calculations using data from ICA Maxi Nacka.
Notes: Numerical values are rounded to the third decimal with p-values reported in square brackets. EQUAIDS 4 were not specified for the regression analyses and are intentionally excluded from the table.

Table A4. Multivariate regression estimates for each level in the demand system

Parameter	Estimate		t-statistic	
	2nd level	3rd level	2nd level	3rd level
γ_{11} {C11}	-.641E-02 (.029)	-.183 (.020)	-.224 [.822]	-8.954 [.000]
γ_{12}{C12}	.0446 (.942E-02)	.028 (.014)	4.736 [.000]	1.940 [.052]
γ_{13}{C130}	.0456 (.864E-02)	.162 (.018)	5.277 [.000]	9.016 [.000]
γ_{22}{C22}	-.0147 (.856E-02)	-.021 (.019)	-1.717 [.086]	-1.108 [.268]
γ_{23}{C23}	-.023 (593E-02)	.014 (.013)	-3.944 [.000]	1.064 [.287]
γ_{33}{C33}	-.824E-02 (.600E-02)	-.192 (.021)	-1.374 [.170]	-9.139 [.000]
β_1{B1}	.011 (.023)	.037 (.014)	.494 [.622]	2.641 [.008]
β_2{B2}	-.026 (.591E-02)	-.030 (.939E-02)	-4.417 [.000]	-3.207 [.001]
β_3{B3}	-.028 (.618E-02)	-.149E-02 (.015)	-4.534 [.000]	-.097 [.923]
α_1{A1}	.577 (.811E-02)	.173 (.494E-02)	71.165 [.000]	35.078 [.000]
α_2{A2}	.083 (.213E-02)	.126 (.326E-02)	39.189 [.000]	38.769 [.000]
α_3{A3}	.072 (.222E-02)	.660 (.533E-02)	32.589 [.000]	123.863 [.000]
μ_1{D1}	.056 (.052)	-.011 (.028)	1.066 [.287]	-.380 [.704]
μ_2{D2}	-.0261 (.014)	-.050 (.019)	-1.907 [.056]	-2.598 [.009]
μ_3{D3}	-.021 (.0143)	.054 (.030)	-1.484 [.138]	1.774 [.076]
Number of obs.	242	242		
Log likelihood	1411.470	1406.000		
Schwarz B.I.C	-1362.060	-1356.600		

Source: Own calculations using data from ICA Maxi Nacka.
Notes: Estimates are rounded to the third decimal with standard errors reported in parentheses and p-values in square brackets. Parameters' TSP notations are reported in curly brackets.

Table A5. Policy impacts under the merged climate footprints approach

SCENARIO 1	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Eco-labelled	0	0	63.93	64.61	1.06	287.13	290.16	3.03	1.06
Bottom trawled	10.27	5.96	42.15	39.35	-6.64	360.65	336.69	-23.95	-6.64
Conventionally produced	0	0	255.33	256.18	0.33	1119.64	1123.36	3.72	0.33
Ecotoxic	0	0	15.59	15.15	-2.78	69.17	67.25	-1.92	-2.78
TOTAL CHANGE	1.26	0.61	377.00	375.29	-0.45	1836.59	1817.46	-19.13	-1.05
Fish	1.26	0.61	377.00	375.29	-0.45	1333.73	1817.46	-19.13	-1.05
Canned Seafood	0	0	49.26	49.29	0.05	241.37	241.50	0.13	0.05
Breaded Seafood	0	0	42.05	42.08	0.05	257.64	257.77	0.13	0.05
Shellfish	0	0	182.09	182.12	0.02	1888.47	1888.81	0.34	0.02
TOTAL CHANGE	0.74	0.38	650.40	648.77	-0.25	4224.06	4205.54	-18.51	-0.44
SCENARIO 2	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Eco-labelled	-20.33	-12.00	63.93	80.37	25.72	287.13	360.98	73.85	25.72
Bottom trawled	10.27	5.96	42.15	37.82	-10.28	360.65	323.59	-37.06	-10.28
Conventionally produced	0	0	255.33	247.27	-3.16	1119.64	1084.28	-35.35	-3.16
Ecotoxic	0	0	15.59	15.34	-1.56	69.17	68.09	-1.08	-1.56
TOTAL CHANGE	-2.18	-1.05	377.00	380.81	1.01	1836.59	1836.94	0.36	0.02
Fish	-2.18	-1.05	377.00	380.81	1.01	1333.73	1836.94	0.36	0.02
Canned Seafood	0	0	49.26	49.21	-0.09	241.37	241.14	-0.23	-0.09
Breaded Seafood	0	0	42.05	42.02	-0.09	257.64	257.41	-0.23	-0.09
Shellfish	0	0	182.09	182.03	-0.03	1888.47	1887.87	-0.60	-0.03
TOTAL CHANGE	-1.28	-0.65	650.40	654.07	0.56	4224.06	4223.36	-0.70	-0.02
SCENARIO 3	ΔP	%ΔP	Q_0	Q_1	%ΔQ	GHG_0	GHG_1	ΔGHG	%ΔGHG
Eco-labelled	0	0	63.93	66.10	3.39	287.13	296.87	9.74	3.39
Bottom trawled	10.27	5.96	42.15	39.64	-5.95	360.65	339.20	-21.45	-5.95
Conventionally produced	5.26	2.32	255.33	249.83	-2.16	1119.64	1095.49	-24.14	-2.16
Ecotoxic	5.33	1.87	15.59	15.18	-2.58	69.17	67.39	-1.79	-2.58
TOTAL CHANGE	4.99	2.40	377.00	370.75	-1.66	1836.59	1798.95	-37.64	-2.09
Fish	4.99	2.40	377.00	370.27	-1.66	1333.73	1798.95	-37.64	-2.09
Canned Seafood	0	0	49.26	49.37	0.22	241.37	241.89	0.52	0.22
Breaded Seafood	0	0	42.05	42.14	0.21	257.64	258.17	0.53	0.21
Shellfish	0	0	182.09	182.22	0.07	1888.47	1889.83	1.36	0.07
TOTAL CHANGE	2.91	1.49	650.40	644.48	-0.91	4224.06	4188.84	-35.22	-0.84

Source: Own calculations using data from ICA Maxi Nacka and climate footprints from Carbon Cloud (2022) as well as Moberg et al. (2019).
Notes: Daily impacts on quantities are expressed in kg, prices in SEK per kg, and GHG emissions in CO_2e.

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