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## LETTER

## The nitrogen footprint of Swedish food consumption

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E-mail: [rasmus.einarsson@chalmers.se](mailto:rasmus.einarsson@chalmers.se) and [christel.cederberg@chalmers.se](mailto:christel.cederberg@chalmers.se)**Keywords:** nitrogen, nitrogen footprint, food system, agriculture, food waste, SwedenSupplementary material for this article is available [online](#)**Abstract**

Food systems are major drivers of environmental and health impacts. While the emissions and other pressures causing these impacts mainly occur in primary agricultural production, the deeper causes and much of the mitigation potential are distributed throughout food systems, including dietary choices and multiple inefficiencies in the whole chain from agricultural production to consumption and waste management. An environmental indicator based on this systems perspective is the nitrogen (N) footprint, defined as the emissions of reactive N due to the consumption of an individual or other entity. Here, we present a method to estimate the N footprint of Swedish food consumption, using a detailed inventory of agricultural production, food and feed processing, food waste, waste management, and wastewater treatment. Limitations of data sources and methods are discussed in detail. The estimated Swedish food N footprint is 12.1 kg N capita<sup>-1</sup> yr<sup>-1</sup>, of which 42% is emitted in Swedish production, 38% in production abroad, 1% in consumer waste management, and 19% in wastewater treatment. Animal food products account for 81% of the food N footprint and 70% of the protein intake. Average protein intake exceeds nutritional requirements by about 60%, which suggests that at least 35% reduction of food-related reactive N emissions could be achieved through dietary change. Of the apparent food N consumption (6.9 kg N capita<sup>-1</sup> yr<sup>-1</sup>), about 22% is food waste N (1.5 kg N capita<sup>-1</sup> yr<sup>-1</sup>). We estimate that 76% of food waste N is unavoidable (bones and other parts not commonly eaten). Avoidable food waste is about 7% of the edible food supply, implying that a hypothetical complete elimination of food waste would reduce emissions by about 7%. In summary, we present a detailed method, discuss its limitations, and demonstrate possible uses of the N footprint as a complement to existing territorial and sectoral environmental indicators.

**1. Introduction**

Food systems are major drivers of environmental and health impacts (Foley *et al* 2005, Springmann *et al* 2018, Gu *et al* 2021). While the emissions and other pressures causing these impacts mainly occur in primary agricultural production, it is increasingly recognized that the deeper causes and indeed much of the potential for mitigation are found in the overall structure of food systems, including dietary choices and multiple inefficiencies in the whole chain from primary production to consumption and waste management (Liu *et al* 2015, Springmann *et al* 2018).

Therefore, in addition to the mostly territorial and sectoral perspective of existing policies for environmental monitoring and mitigation, there is now a strong interest in information and policies that explicitly address the whole food system, connecting food consumption to all its direct and indirect effects on environmental quality and human health and well-being (Willett *et al* 2019, Kugelberg *et al* 2021). A recent example is the EU Farm to Fork strategy (European Commission 2020), which asserts that a 'shift to a sustainable food system can bring environmental, health and social benefits, offer economic gains and ensure that the recovery from

the [COVID-19] crisis puts us onto a sustainable path’.

This paper is about the emissions of reactive nitrogen (N) due to Swedish food consumption. N is a major driver of agricultural productivity but unfortunately also a common element of several pollution streams contributing to a range of environmental problems including eutrophication, acidification, air pollution, climate change, and biodiversity loss (Sutton *et al* 2011). A food system perspective is needed to efficiently combine the multiple levers for mitigation distributed throughout the food system (Kanter *et al* 2020, Leip *et al* 2020, Uwizeye *et al* 2020).

Sweden, a high-income country with a population of 10 million and moderately intensive agriculture on about 3 million hectares of agricultural land, is a net importer of food and therefore also causes substantial food-related environmental impacts abroad (Cederberg *et al* 2019). Swedish environmental policy acknowledges the importance of consumption-based environmental accounting in its overarching Generational Goal, which calls for solving the major environmental problems in Sweden within one generation without causing additional environmental and health impacts outside Sweden’s borders (Swedish Environmental Protection Agency 2018). Major efforts to quantify some of these impacts have recently been made in the project Policy Relevant Indicators for Consumption and Environment (PRINCE, [www.prince-project.se](http://www.prince-project.se)). Specifically, the PRINCE project quantified consumption-based indicators of air pollution, greenhouse gas emissions, land use, materials consumption, blue water consumption, and use of hazardous chemicals (Cederberg *et al* 2019, Palm *et al* 2019, Persson *et al* 2019).

So far, however, no comprehensive assessment has been made of the reactive N emissions due to Swedish food consumption. While the previously mentioned PRINCE project made a preliminary analysis of food-related N emissions, suggesting that likely more than half of the reactive N emissions due to Swedish food consumption in 2011 occurred outside the country’s borders, it was pointed out in the project report that these results are highly uncertain and that there is a need for further method development (Steinbach *et al* 2018). In this study, we estimate the N footprint of Swedish food consumption. The N footprint is defined as the total emissions of reactive N to the environment due to the consumption of an individual or other entity (Leach *et al* 2012, Leip and Uwizeye 2019). Several estimates of food N footprints have been made globally and in Europe, for example in Austria (Pierer *et al* 2014), EU27 (Leip *et al* 2014), Germany (Klement *et al* 2021), Portugal (Cordovil *et al* 2020), and the United Kingdom (Stevens *et al* 2014). This paper adds Sweden to the growing list of European countries where methods and data sources to estimate N footprints have been

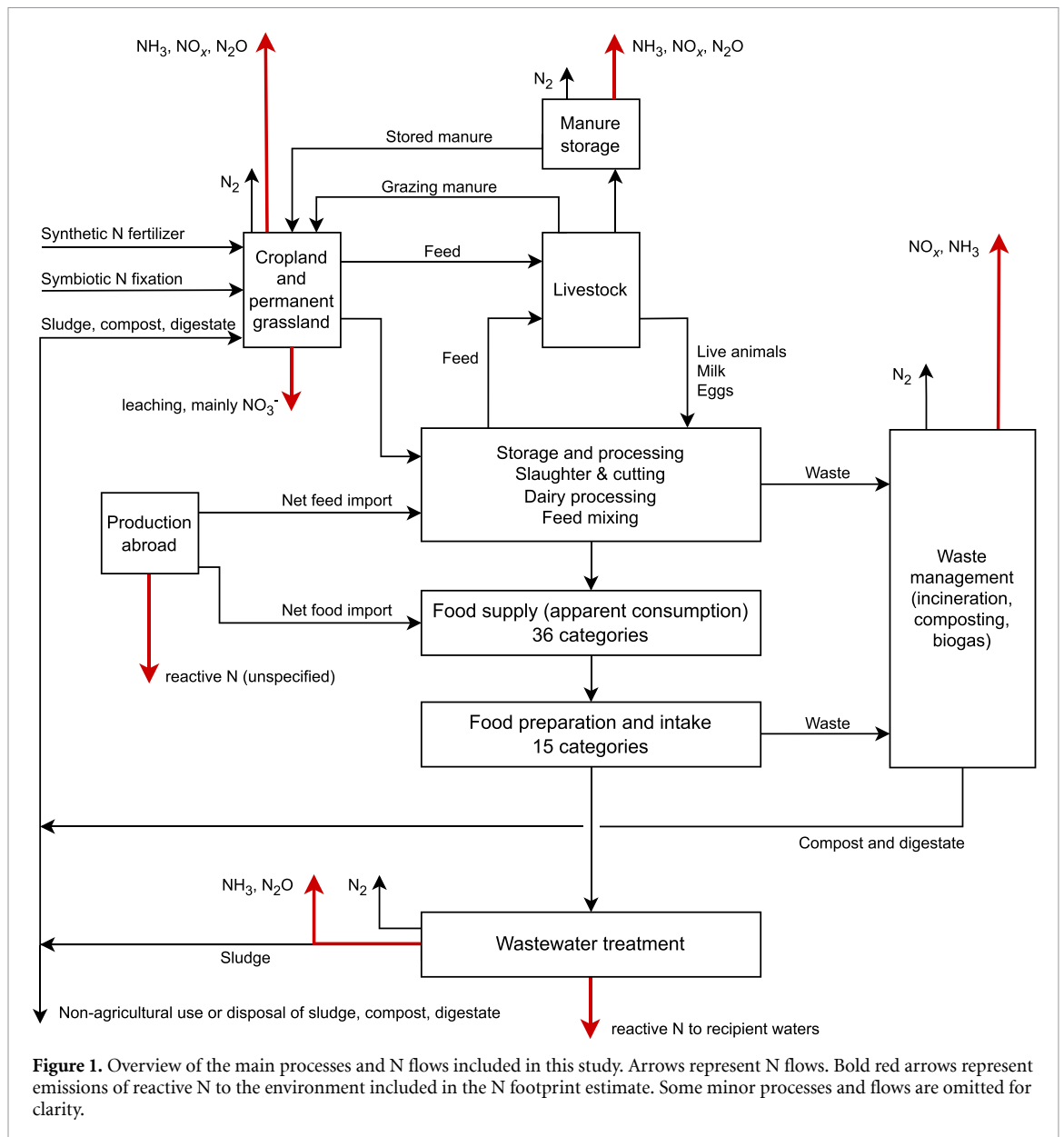
explored. Our N footprint estimate accounts for N emissions embodied in international food and feed trade, thus quantifying the division of N emissions within and outside Sweden’s borders.

What functions can the N footprint have as a complement to the emission inventories and other quantitative indicators already used in various environmental policies? In Sweden, some examples of policies that monitor emissions of reactive N are the Gothenburg Protocol of the Convention on Long-Range Transboundary Air Pollution (CLRTAP), the UN Framework Convention on Climate Change (UNFCCC), and the Helsinki Convention. The emission inventories and other indicators compiled within these policies are typically reported by national territory, economic sector, chemical form, and/or recipient environment, and thus provide a rich picture of the progress towards national policy targets. By comparison, the N footprint is clearly a weaker predictor of specific environmental and health risks, but its consumption perspective and its relative simplicity may prove useful to create an overview of the contributions of different pollution streams throughout the food system, the contributions of different food categories, and the overall share of consumption-driven N pollution occurring outside Sweden’s borders.

This paper makes three main contributions. First, we develop a method to estimate the N footprint of Swedish food consumption. In order to maximize transparency, consistency with official statistics, and the possibility to make periodic updates, the method is based as far as possible on publicly available statistics and accounting methods used in national and international reporting (e.g. to the UNFCCC, CLRTAP, and Helsinki Convention). Our inventory of primary production, food and feed processing, apparent consumption, food waste, waste management, and wastewater treatment is highly detailed and can be used for a wide range of analyses of the Swedish food system. Second, based on the main challenges encountered in the method development, and considering envisioned uses of the N footprint, we suggest some potential improvements to data sources and methods that should be given priority if the N footprint is to be used as one among several tools to monitor the N pollution due to the Swedish food system. Third, we demonstrate how the N footprint can be partitioned in different ways to provide quantitative perspectives on dietary choices, food waste, wastewater treatment, food production systems, international trade, and national food self-sufficiency.

## 2. Method

This paper presents a method to estimate the N footprint of Swedish food consumption, covering the whole food system including production of synthetic fertilizer, crop production, livestock production, food and feed processing, retail, food waste, solid waste



management, and wastewater treatment. The food N footprint, defined as the reactive N emissions to the environment due to food consumption (Leach *et al* 2012, Leip and Uwezeye 2019), is here calculated as the sum of major food-system emissions of reactive N (see figure 1). Emissions of unreactive  $N_2$  from wastewater treatment, manure management, agricultural soils, and waste combustion do not contribute to the N footprint presented here. Flows of N in the food system were traced to the extent needed to establish the N footprint of Swedish food consumption. Hence, non-food products such as biofuels were not included except for the purpose of determining the emission intensity of feed by-products such as spent grain from biofuel production. Where possible, we used average data for the years 2015–2017.

Here, we give an overview description of our methods and data sources. We also provide supplementary materials (SM) including an in-depth

description of data and methods (SM1) as well as an Excel file with the full calculation and all results (SM2). Figure 1 and table 1 provide an overview of the methods and key data sources used in this study.

### 2.1. Apparent food consumption, food waste, and actual food intake

To estimate the per-capita apparent food consumption, we mainly used official Swedish statistics on ‘total consumption’, defined as the total quantity of different food items supplied to food industry, households, and restaurants (Swedish Board of Agriculture 2021d). Examples of items in the total consumption statistics are flour, rice, potatoes, vegetables, fruit and berries, meat (as carcass weight), various dairy products, eggs, oils and margarine, etc. In addition to the total consumption statistics, we used separate data sources on consumption of alcoholic beverages (Swedish Board of Agriculture 2021c),

**Table 1.** Overview of methods, assumptions, and key data sources. For full detail, see SM1 and SM2.

Methods and assumptions	Key references
<i>Food consumption, food waste, and related emissions</i>	
Apparent consumption, i.e. food supply to households and restaurants, determined in 36 categories, primarily using national consumption statistics.	Swedish Board of Agriculture (2021d), Borthwick <i>et al</i> (2019).
Food intake calculated as apparent consumption minus food waste in households and restaurants. Food waste estimated for 15 aggregated food categories based on national surveys, mass balances, and research literature. Waste management and related N emissions accounted for. Wastewater treatment and related emissions based on national statistics.	Andersson (2012), Andersson and Stålhandske (2020), Gustavsson <i>et al</i> (2011, 2013), Le Noë <i>et al</i> (2017). Statistics Sweden (2018).
<i>Food and feed processing</i>	
Processing and waste in food and feed industry accounted for using mass balances and economic allocation. Examples of processes include milling of cereals and oilseeds, sorting and storage losses of fruit and vegetables, fermentation of alcoholic beverages, slaughter and cutting, and dairy processing. Waste management and related N emissions accounted for.	Flysjö <i>et al</i> (2008, 2014), Gustavsson <i>et al</i> (2013), Le Noë <i>et al</i> (2017). See also SM.
<i>International trade</i>	
Net food imports calculated on the level of the 36 categories of apparent consumption using official trade statistics. Net feed imports estimated using trade statistics and feed industry data.	Swedish Board of Agriculture (2016, 2018a, 2018b), Statistics Sweden (2021).
<i>Primary crop products</i>	
Productivity and emission intensity estimated in 35 crop categories produced in Sweden and abroad, primarily by dividing emission estimates by production quantities. Some data gaps filled using soil surface N budgets.	—
Emissions of NH <sub>3</sub> , N <sub>2</sub> O, NO <sub>x</sub> , and leaching (mainly NO <sub>3</sub> <sup>-</sup> ) calculated using national crop-specific statistics on fertilizer and manure use, and emission estimation methods following national reporting to CLRTAP, UNFCCC, and the Helsinki Convention. Emission estimates cross-checked against soil surface N budgets and broadly found in agreement.	Statistics Sweden (2017), Swedish Environmental Protection Agency (2019a, 2019b), Johnsson <i>et al</i> (2019).
Cropland production in Sweden based on national statistics. Temporary and permanent grassland production based on national statistics in combination with a detailed feed supply balance.	Swedish Board of Agriculture (2021a, 2021b), Cederberg and Henriksson (2020).
<i>Primary livestock products</i>	
Emission intensities for beef, lamb, pork, poultry meat, milk, and eggs calculated based on detailed inventory of Swedish production volumes, feed use, and emissions from stables and manure storage.	—
Feed supply to Swedish livestock estimated for 39 feed items representing eight feed categories: silage and hay, pasture, grain, protein feeds, fat, by-products, dairy products and by-products, Minerals and other. A range of data sources were combined and cross-checked to produce estimates consistent with national statistics on crop production and trade, livestock populations, and expert estimates of livestock metabolic requirements.	Swedish Board of Agriculture (2016, 2018a, 2018b), Cederberg <i>et al</i> (2009), Cederberg and Henriksson (2020), Landquist <i>et al</i> (2020).
Emissions of reactive N from stables and manure storages were calculated following national reporting to CLRTAP and UNFCCC.	Swedish Environmental Protection Agency (2019a, 2019b).

pulses (FAOSTAT 2020), and seafood (Borthwick *et al* 2019).

The combined dataset on apparent consumption contains 36 food categories (see SM2) which we aggregated to 15 broader categories: cereals, potatoes, sugar, fruit and vegetables, pulses, margarine and vegetable oils, alcoholic beverages, beef, lamb, pork, poultry meat, offal and other meat, dairy, eggs, and seafood.

The apparent consumption is larger than the actual food intake due to the waste occurring in food

industry, households, and restaurants. We estimated the unavoidable and avoidable waste between apparent consumption and actual food intake using several Swedish data sources (Andersson 2012, Brancoli *et al* 2017, 2019, Andersson and Stålhandske 2020; see the SM for details). Unavoidable food waste is here defined as the parts of the total consumption normally not intended for human consumption and therefore removed in food preparation, such as bones, egg shells, and fruit and vegetable parts such as peels and kernels. Avoidable food waste is the remaining

waste. While we recognize that the division between unavoidable and avoidable food waste is culturally determined and also rather uncertain, we nevertheless find the division useful to give an order-of-magnitude estimate of how much food waste could be reduced by adopting less wasteful practices for storage and handling. Food waste N is mainly treated in two ways. Either it is incinerated, in which case most N is transformed to  $N_2$  (cf Esculier *et al* 2019); or it is composted or anaerobically digested and subsequently used as fertilizer (Andersson and Stålhandske 2020). Direct emissions related to waste management are therefore small (perhaps 3%; see the SM for details). Later emissions occurring during fertilizer use were allocated to crops (see section 2.4).

In Sweden, about 87% of human N excretion is treated in municipal wastewater treatment plants. The remainder is divided between various types of privately owned collection or treatment systems. Based on statistics about wastewater treatment efficiencies and sludge use (Ek *et al* 2011, Olshammar 2017, Statistics Sweden 2018) we estimated that about 44% of human N excretion is emitted to the environment (mainly recipient waters) after treatment. See SM1 and SM2 for further details.

## 2.2. Trade balance

To estimate the shares of N emissions occurring in Sweden and abroad, we established a simple trade model on the level of the 36 consumption categories. We determined the net trade balance in each category, primarily using official trade statistics (Statistics Sweden 2021). In categories with net imports, we calculated the import share as net import divided by total consumption. In categories with net export, we set the import share to zero. For wheat and rye flour, beer, and margarine, we accounted for the trade balance in terms of primary crop products (wheat, rye, barley, and oilseeds); see SM1 section 2.2.3 for details. Moreover, we accounted for feed imports to Swedish livestock production as detailed in section 2.5 below.

## 2.3. Processing of primary crop and livestock products into food and feed

Many products in the food system are intrinsically linked as co-products. For example, wheat flour is co-produced with wheat bran, rapeseed oil is co-produced with rapeseed meal, meat is co-produced with offal, and whole milk is used to produce cheese, cream, and other dairy products.

For most of these co-production processes, we divided the upstream nitrogen emissions among co-products using economic allocation. Conversion ratios and waste production as well as allocation factors were determined using various literature sources, and in some cases additional assumptions. See the SM for full details.

## 2.4. Crop production

For primary crop products used in food and feed production, we estimated N emissions including leaching and runoff as well as gaseous emissions related to fertilizer application and crop residues. Gaseous N emissions related to manure application and livestock excretion on pasture were allocated to the crop products, whereas N emissions from stables and manure storage were allocated to the livestock products (see section 2.5 below).

For crop production in Sweden, we estimated emissions following the methods used in Sweden's national emission inventories within the UNFCCC, CLRTAP, and Helsinki Convention (Johnsson *et al* 2019, Swedish Environmental Protection Agency 2019a, 2019b), in combination with crop-specific data on areas, harvests, and N inputs (synthetic N fertilizers, manure, sludge from wastewater treatment, and other biologically treated waste). See the SM for details.

Another common method for estimating N emission intensities for crop products is to use N surpluses calculated from soil-surface N budgets. For comparison and validation, we therefore also established N budgets for all the Swedish crops and compared the surpluses to the direct estimates of emissions. Key input data to the budgets were national crop-specific statistics on N inputs from synthetic fertilizers and manure, which was cross-checked for agreement with total national manure supply; and a detailed national model for symbiotic N fixation in grass-clover leys. While the emission estimates broadly agree (figure SM5), there are also a few considerable discrepancies which may have several causes as further discussed in SM1.

For imported crop products, we estimated N footprints in one of two ways. For crops that are also cultivated in Sweden (e.g. rapeseed, sugar beets, potatoes), we assumed that the N footprint of production abroad is equal to the Swedish production. For a small number of other crops (e.g. rice, soybeans, some fruits and vegetables), we used a simplified emission estimate based on soil-surface N budgets (see SM for details).

## 2.5. Livestock production

We estimated emission intensities of primary livestock products including upstream N emissions in crop production and feed processing as well as direct emissions from manure in stables and manure storage systems.

For livestock production in Sweden, we made detailed estimates of feed use and manure management and related emissions, accounting also for the emissions abroad due to imported feed. For imported animal products, we assumed equal emission intensities as in Swedish production, and that all the related emissions occur outside Sweden. The remainder of

this section outlines how we estimated emissions related to Swedish livestock production.

We calculated the emission intensities of the primary livestock products by dividing the total Swedish emissions in each livestock category by the total Swedish production of the corresponding product. For milk and beef, which are co-produced in dairy systems, we allocated emissions as follows. The milk was allocated feed and manure emissions from dairy cows and replacement heifers and calves; and the beef (including meat from the dairy herd) was allocated all other feed and manure emissions from bovine livestock, i.e. the suckler beef herd as well as surplus dairy calves raised for meat.

Feed consumption and related emissions were estimated for each primary livestock product in 39 feed items aggregated to eight categories (silage and hay, pasture, grain, protein feeds, fat, by-products, dairy products, and minerals and other) using a variety of data sources, including an inventory of roughage feed production and use (Cederberg and Henriksson 2020), life-cycle inventories of pig and poultry production (Cederberg *et al* 2009, Landquist *et al* 2020), national data on concentrate feed, and expert estimates of feed rations (see SM for details). For feed categories with net imports (most importantly protein feeds based on soy and rapeseed) we determined feed import shares similar to the trade balance for food consumption (section 2.2) and accounted for the corresponding division of emissions between Sweden and the rest of the world.

Silage feed is an important feed component for Swedish ruminants. We accounted for the losses of dry matter from silage production systems, which are mainly gaseous and can be substantial (1%–14%) depending on feedstock and storage systems (Spörndly 2018, Cederberg and Henriksson 2020). In addition to the relatively well-known losses of dry matter and feed energy (Borreani *et al* 2018), recent research also suggests that substantial quantities of N are lost in these processes (Spörndly 2018, Köhler *et al* 2019). The N losses likely escape mainly as reactive N but also partly as N<sub>2</sub> (Spörndly *et al* 2021). We included an estimate of these reactive N emissions as further explained in the SM.

N emissions from stables and manure storage were estimated using the same methods used for Swedish emission inventories to the CLRTAP and UNFCCC (Swedish Environmental Protection Agency 2019a, 2019b).

### 2.6. Alternative estimate of emissions from imported food

N emissions in food production vary between countries. In this study, however, we have approximated the emission intensities of imported food and feed as equal to emission intensities of corresponding Swedish products. There are two main reasons for this method choice. First, the complexity of global supply

chains makes it difficult to know where imported products were produced. As an example, imported meat might be imported to Sweden from Germany, even if the livestock was reared in the Netherlands, using feeds from yet another country, and so on (see, e.g. Kastner *et al* 2011, Wood *et al* 2015, Bruckner *et al* 2019). Second, even if the full supply chains were known, existing datasets of food N footprints from various countries are yet so diverse in terms of methods (system boundaries, treatment of co-products, and other assumptions) that differences in results between countries may equally well be caused by differences in assessment methods as in actual emission intensities (Einarsson and Cederberg 2019).

Nevertheless, to give an idea of how different emission intensities of imported food could affect the results, we calculated two alternative estimates of food N footprints using virtual N factors (VNFs) (Leach *et al* 2012) estimated in previous research. The VNF is the emission intensity for a food item, counting only the emissions occurring before consumption (Leach *et al* 2012). We used VNFs from a mix of European studies (Shibata *et al* 2017) and from a recent German study (Klement *et al* 2021) because most food imported to Sweden is produced in Europe (Cederberg *et al* 2019).

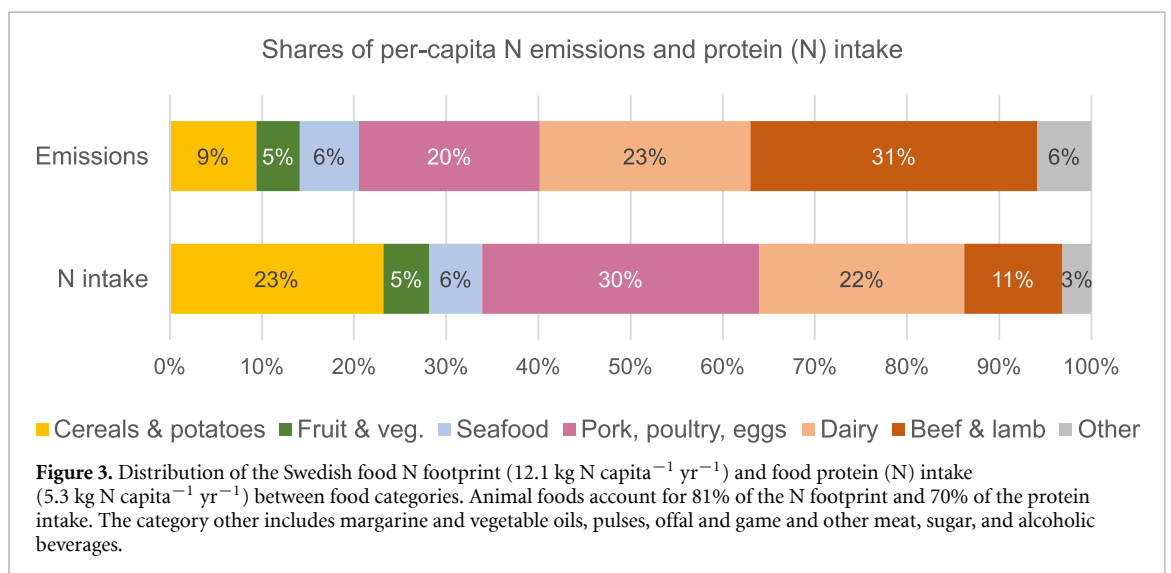
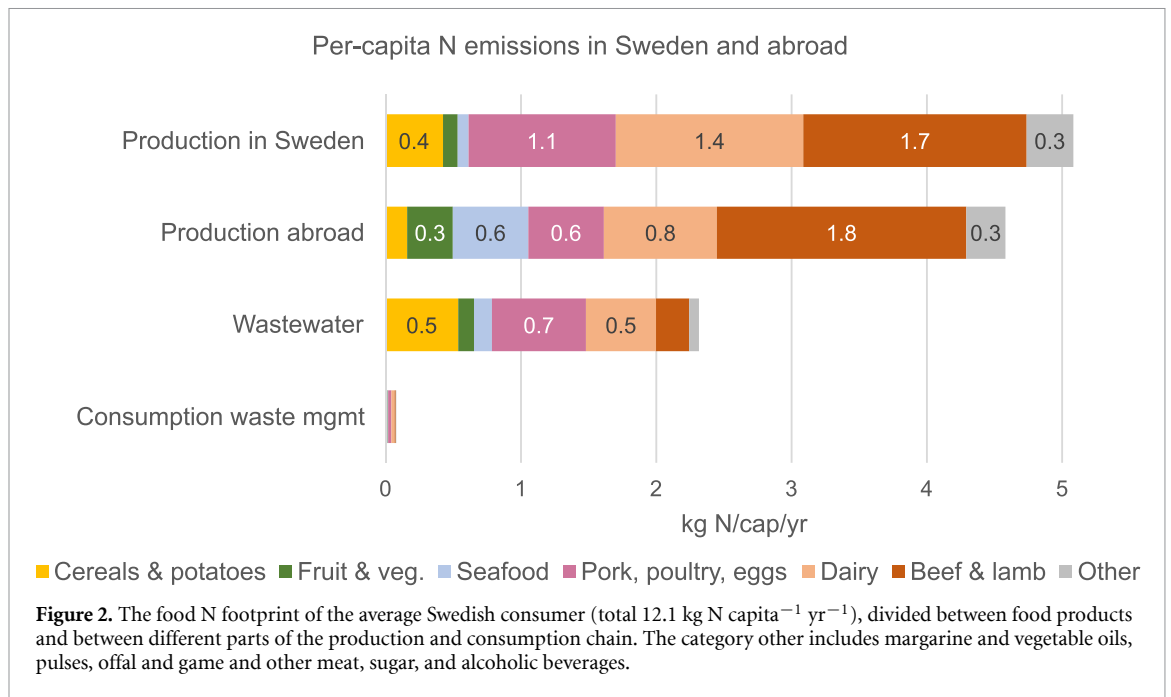
## 3. Results

The N footprint of Swedish food consumption in 2015–2017 was 12.1 kg N capita<sup>-1</sup> yr<sup>-1</sup>. About 81% of emissions occurred before human intake (i.e. in agricultural production, processing, and food waste management) and the remaining 19% in wastewater treatment (see figure 2).

Emissions occurring abroad accounted for 38% of the total Swedish food N footprint, and 47% of production-related emissions (see figure 2). Sweden imported about 30% of food N, calculated at the level of total consumption. However, the share of N emissions occurring abroad was larger since imports were dominated by relatively emission intensive foods: animal products accounted for 78% of food N imports, and fruit and vegetables for 13%.

When replacing our estimates of VNFs (pre-consumption emission factors) of imported food by VNFs from other European studies, the total N footprint of Swedish food consumption increased by 24% when using the Europe average from Shibata *et al* (2017) or 1% using the results from Germany of Klement *et al* (2021); see also figures SM9 and SM10.

Animal foods accounted for about 70% of N intake and 81% of N emissions (figures 3 and 4). The higher share of animal foods in N emissions than N intake reflects the generally higher N emission intensity of animal protein sources: the VNF, expressed as a dimensionless ratio of N emissions to N intake, was more than twice as high for animal products (2.2) compared to vegetal products (1.0).

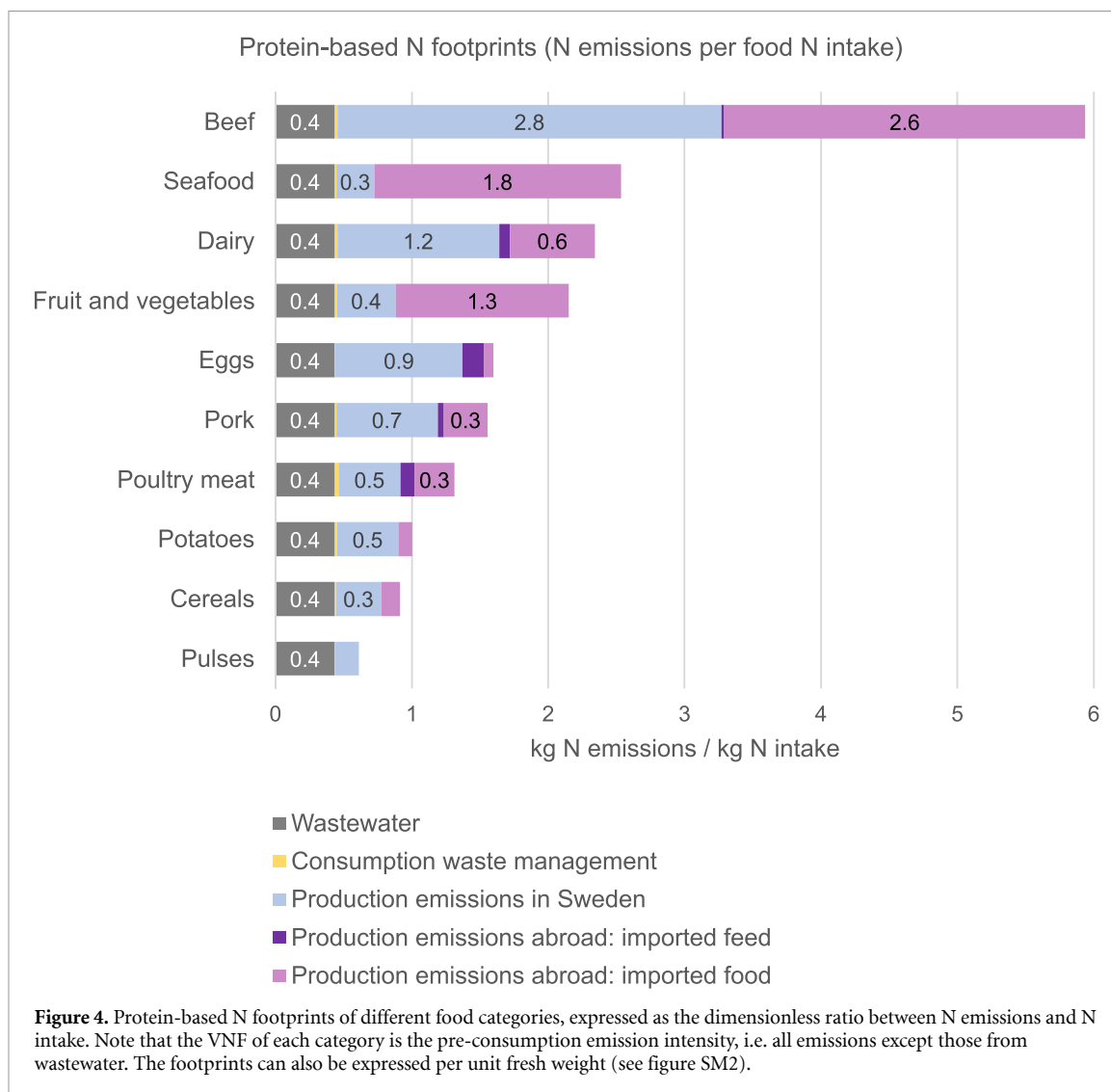


Within both these categories, however, there were substantial variations (see figure 4). For example, the VNF was much higher for beef (5.5) than dairy products (1.9), eggs (1.2), pork (1.1), or poultry meat (0.8). Among vegetal products, VNFs were relatively low for pulses (0.2), cereals (0.5), and potatoes (0.6), and relatively high for fruit and vegetables (1.7). VNFs can also be expressed per unit fresh weight of the food categories (see figure SM2).

The main explanation for differences in VNFs between animal products is the differences in feed consumption, and therefore also manure excretion, per unit edible product. Higher feed consumption results in higher emissions from both feed production and manure management. The dimensionless ratio of N in feed intake to N in human-edible product varied as follows: for beef 15.8, for whole milk 5.4, and

for pork, poultry meat, and eggs 3.2–3.6 (see figure SM7a). While individual feed products have different emission intensities, the overall feed mixes for different animal products were all rather similar in N emission intensity (see figures SM7 and SM8). The estimated gaseous emissions of reactive N from silage production account for 8% and 12% of beef and dairy VNFs, respectively. Another partial explanation for differences in VNFs is that different manure management systems are used (e.g. relatively more liquid manure in dairy production), resulting in different shares of livestock N excretion emitted to the environment (see figure SM6).

The total consumption of food N was 6.9 kg N capita<sup>-1</sup> yr<sup>-1</sup> while the actual intake was 5.3 kg N capita<sup>-1</sup> yr<sup>-1</sup>. The difference, i.e. food waste primarily in restaurants and households, was



in terms of N dominated by unavoidable waste ( $1.2 \text{ kg N capita}^{-1} \text{ yr}^{-1}$ , 76% of food waste N), mainly bones (see figures 5 and SM11). The avoidable food waste N,  $0.4 \text{ kg N capita}^{-1} \text{ yr}^{-1}$ , was 53% animal N. In terms of fresh weight, the majority of waste was fruit and vegetables (43% of waste fresh weight), followed by potatoes (18%). In total, vegetal products accounted for 71% of total waste fresh weight but only 20% of total waste N.

## 4. Discussion

### 4.1. Comparison to previous N footprint estimates

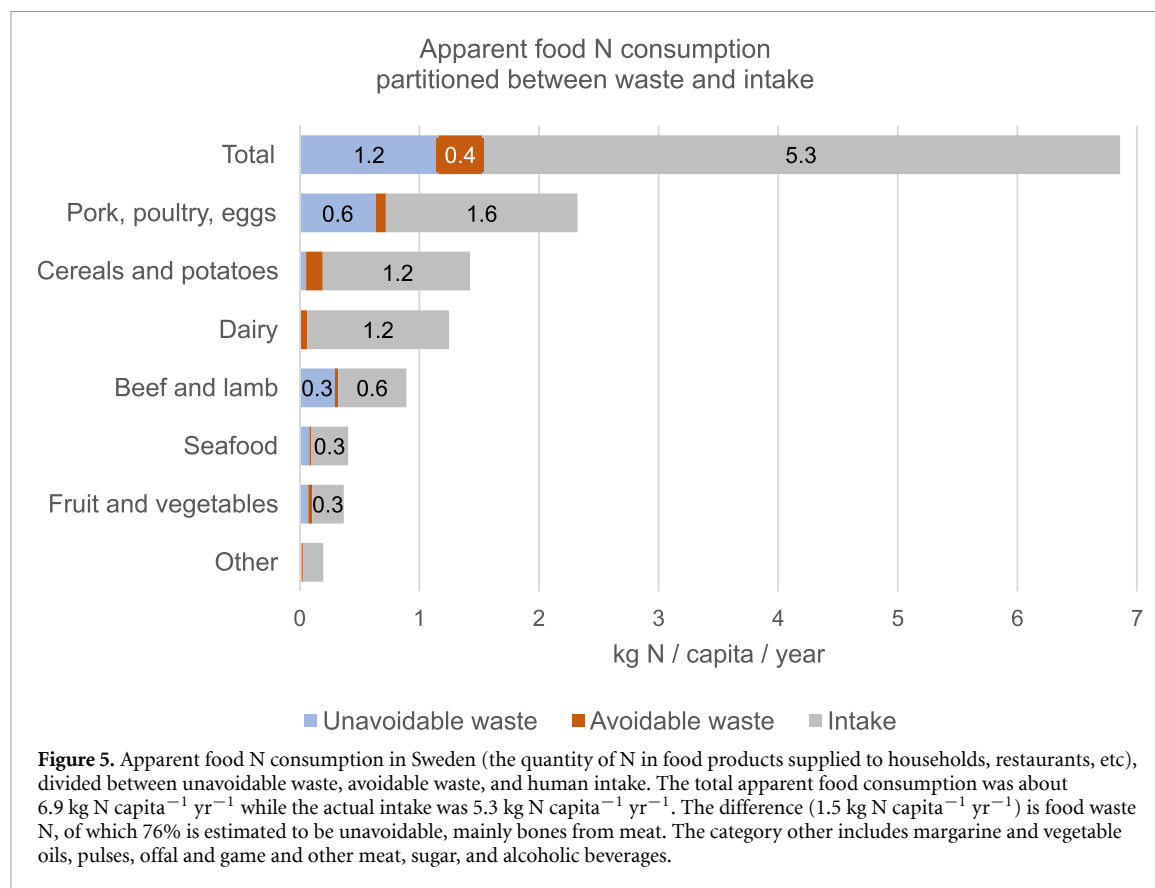
Direct comparison of N footprint estimates between studies is not straightforward. In addition to actual differences in diets and production systems, results may also be substantially affected by method differences such as system boundaries, trade models, allocation methods, and data sources (Einarsson and Cederberg 2019). In summary, we believe that differences between our VNF estimates and those of other studies in some cases may equally depend on differences in methods as in differences in actual

underlying N flows. Considering this, we found it particularly relevant to compare results with the German study by Klement *et al* (2021) since the methods and level of detail in that study are similar to ours, yet based on data from German production systems (see Klement *et al* 2021, supplementary information section 7, pp 29–32). While VNFs for individual food items differ considerably in some cases (see figure SM10), the diet-weighted totals are similar (section 2.6 and figure SM9). There is no conclusive evidence that N emission intensity of German food production on average is different from that of Swedish production. Further research efforts would be needed to draw more specific conclusions about the comparability of these two studies.

### 4.2. Possible improvements in data and methods

Here, we highlight four areas with room for improvement in data and methods.

First, the modeling of international trade and production systems is a major challenge in environmental assessments of food products both due to conceptual challenges related to complex trade flows and



due to limited data about trade and production systems. The physical trade model used in this paper has distinct advantages in terms of simplicity and good data coverage on the level of the 36 food consumption categories. However, our trade model also has limitations in terms of consistency and comprehensiveness compared to input–output models and similar approaches which explicitly and consistently model different production systems and international trade flows in economic and/or physical terms (see, e.g. Kastner *et al* 2011, Wood *et al* 2015, Oita *et al* 2016, Bruckner *et al* 2019, 2021). Comparative evaluation of these different approaches is beyond the scope of this paper, but we wish to emphasize the importance of carefully considering the strengths and weaknesses of different trade models in any further development of consumption-based indicators such as the food N footprint.

Second, food waste throughout food systems is subject to a growing interest in both research and policy. While methods and data for measurement and reporting are improving both internationally and nationally (e.g. WRI 2016, Lindow *et al* 2021, Östergren *et al* 2021), there is nevertheless a lack of detailed, reliable, and up-to-date information about the types and quantities of food waste occurring at different stages. In this paper, we have estimated unavoidable and avoidable food waste focusing mainly on the retail and consumption stages, using research results and official statistics from Sweden.

Further efforts, such as those ongoing in Swedish industry and government agencies (e.g. Lindow *et al* 2021, Östergren *et al* 2021), will be very useful to further improve the accuracy and detail of food waste estimates.

Third, estimates of livestock feed production, trade, waste, and use for different livestock categories are of crucial importance for our results, considering the large share of the Swedish food N footprint related to animal products. Unfortunately, national feed consumption is not regularly measured or estimated and therefore the results presented here are based on a mix of data sources including feed trade data (collected for feed safety purposes), crop supply balances, and expert estimates. Despite considerable efforts with these estimates, our impression is that they remain rather uncertain and that additional data collection and/or further efforts with systematic and recurring estimates would be very valuable to improve the accuracy and level of detail of Swedish feed consumption data.

Finally, it must be noted that the food N footprint is a rather simplistic indicator of the multi-dimensional and localized environmental and health impacts of food-related N pollution (Einarsson and Cederberg 2019). While our results here agree with previous studies (e.g. Leach *et al* 2012, Shibata *et al* 2017, Klement *et al* 2021) on the large differences between different food categories and thus give useful information about the mitigation potential of dietary

change, there remains a need for research on (a) how N footprints correlate with more specific impact indicators, and (b) based on this, what role the N footprint can usefully play in guiding policy towards desirable outcomes.

### 4.3. Insights gained by disaggregating the N footprint along multiple axes

Our detailed treatment of N flows throughout the food system enables disaggregation of the per-capita food N footprint along multiple axes. For example, the N footprint can be divided between food categories, between different parts of the production and consumption chain, or between Sweden and the rest of the world. Here, we discuss a few examples of such disaggregations and some insights that can be gained from them.

What is the potential to reduce the Swedish food N footprint through dietary change? While a rigorous treatment of this question would require further research, some preliminary analysis focusing on protein supply can already be made. Since food categories vary widely in protein-weighted N footprint (figure 4), there is a substantial potential to reduce the Swedish food N footprint by reducing the consumption of high-footprint protein sources, replacing them partially or fully by lower-footprint protein sources. The average protein intake in Sweden is here estimated at about 90 g protein capita<sup>-1</sup> d<sup>-1</sup> (5.3 kg N capita<sup>-1</sup> yr<sup>-1</sup>), exceeding adult nutritional protein requirements of about 56 g protein capita<sup>-1</sup> d<sup>-1</sup> (WHO 2007) by about 60%. Thus, considering only protein supply, it would be possible to reduce protein intake by about 35%, and if high-footprint protein sources were preferentially reduced, this would entail a total food N footprint reduction of more than 35%. While further research into this question should also consider nutritional adequacy besides total protein supply, the magnitude of this result suggests that dietary change is a major possibility to reduce the Swedish food N footprint.

What is the mitigation potential in minimizing food waste? The answer to this question can be divided into the potential to reduce (a) direct emissions related to food-waste management and (b) 'upstream' emissions related to primary production of those crop and livestock products that are subsequently removed as avoidable food waste. The first category, according to our results (and those of Esculier *et al* 2019), is very minor (see sections 2.1 and SM), in contrast to some other N footprint estimates where all or a large share of food waste N is assumed emitted to the environment as reactive N (e.g. Leach *et al* 2012, Pierer *et al* 2014, Klement *et al* 2021). The second category is more substantial. Our results (figures 5 and SM) show that avoidable food waste amounts to about 7% of the food supply after unavoidable waste; and consequently, a similar share of all upstream production emissions could

hypothetically be avoided by completely eliminating avoidable food waste while keeping food intake constant. We conclude that realistically attainable reductions in avoidable food waste (perhaps 50%–75%; see Springmann *et al* 2018) could at most reduce the Swedish food N footprint by perhaps 3%–5%. An important caveat to this result is that it hinges on the division between avoidable and unavoidable waste, which is culturally determined. The mitigation potential calculated here is the N emissions that could be avoided without changing the quantity or types of food eaten. See SM1 for further detail.

What is the mitigation potential inside and outside Swedish territory? Our results show that about 30% of Swedish food N is imported, and about half of the production-related (i.e. mainly agricultural) emissions occur outside Sweden's borders (figure 2). This perspective is a valuable complement to the territorial perspective taken by national emission inventories (e.g. within UNFCCC, CLRTAP, and the Helsinki Convention). The considerable magnitude of the N emissions embodied in imported food underscores the need for continued and improved monitoring of trade-related emissions in order to ensure that Sweden's Generational Goal is reached.

## 5. Conclusion

This paper presents an estimate of the Swedish food N footprint based on a detailed inventory of Swedish food production and consumption. The N footprint is a consumption-based indicator, complementing the territorial perspective taken by most indicators used in environmental policy, and encouraging a systems-oriented understanding of emissions and mitigation potentials throughout the food system. As we have demonstrated, the food N footprint can be partitioned in different ways to provide quantitative perspectives on dietary choices, food waste, wastewater treatment, food production systems, international trade, and national food self-sufficiency. Despite some limitations in data and methods, which we have discussed in detail in this paper and the SM, we find the food N footprint presented here a useful tool among others to understand the potential to mitigate the environmental and health impacts of the Swedish food system.

### Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files).

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