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Decreasing dietary nitrogen consumption improves wastewater treatment efficiency and carbon footprint

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ABSTRACT

This article aimed to connect protein consumption with the nitrogen load to wastewater treatment plants (WWTPs) in Finland. The influence of the changes in nitrogen consumption on the WWTP environmental footprint was estimated using process simulation. As the main result, a connection was found between nitrogen loads from food consumption and the incoming load to a WWTP. This was done by analysing protein consumption data from the Food and Agriculture Organization of the United Nations (FAO) and incoming nitrogen load data from the Finnish environmental institute, SYKE. The impact of nitrogen consumption was estimated using different diet scenarios. Decreasing dietary nitrogen consumption by 16–24% could decrease nitrous oxide emissions by 16–24% and aeration energy (AE) consumption by 6–11%. An increase in dietary nitrogen consumption of 6–42% could increase AE consumption by 2–14% when effluent requirements were met. When considering the environmental impact of this increased aeration, it corresponds to an increase of 2–16%. Furthermore, nitrous oxide emissions could rise by 6–42% This information can be valuable to WWTPs and even consumers for influencing incoming nitrogen loads.

Key words: dietary nitrogen consumption, environmental impacts, greenhouse gas emissions, nitrogen load to water bodies, nitrogen removal, wastewater treatment

HIGHLIGHTS

- Dietary choices affect the nitrogen load to the wastewater treatment.
- Protein consumption and nitrogen load to the wastewater treatment have a strong correlation.
- Recommended diets would decrease the environmental impact of wastewaters.
- Nitrogen load strongly influences the climate impact of wastewater treatment.

INTRODUCTION

Personal food consumption has rarely been connected to wastewater quality or wastewater treatment plant (WWTP) performance. A study in the USA claims that matching human consumption to protein requirements would reduce nitrogen losses to aquatic ecosystems (Almaraz *et al.* 2022). The nutrient loads to municipal WWTPs are mainly anthropogenic or human sources such as human faeces and urine or from household activities, with the major part of nitrogen (75%) coming from urine (Tjandraatmadja & Diaper 2006). Antikainen (2007) also states that the overall majority of consumed nutrients end up in household wastewater.

Removal techniques are developing constantly, but nutrient loads to WWTPs also have a tendency to increase (Säylä 2015). Nutrient overload to the water bodies causes eutrophication, which can lead to algal blooms, aquatic hypoxia and anoxia. Population and economic growth in developing countries will lead to an increasing amount of nitrogen in the sewer systems, which can lead to severe eutrophication (Van Drecht *et al.* 2009). It has been estimated that 10% of the nitrogen (N) load into the Baltic Sea is from municipal WWTPs (Mörth *et al.* 2007). Van Drecht *et al.* (2009) assume human N emissions to be dependent on income level.

Dietary nitrogen is mostly included in the protein content of food. Nitrogen is also found in other foodstuffs, but their share is less significant (Mariotti *et al.* 2008). Therefore, nitrogen ends up in the WWTPs through consumed protein. The average nitrogen concentration of protein is 16%, which usually varies between 13 and 19% (FAO 2003). The main protein sources are meat and milk products, fish and eggs, and plant-based sources such as pulses, seeds, nuts and different soy products (THL 2021). Meat consumption is studied as a food with high nitrogen content (Leach *et al.* 2012; Hayashi *et al.* 2018; Martinez *et al.* 2019).

According to the Intergovernmental Panel on Climate Change (IPCC) (2015), waste and wastewater treatment cause about 3% of all global greenhouse gas (GHG) emissions. In Maktabifard *et al.*'s (2022) study, direct emissions are considered emissions from treatment and sludge treatment processes. Energy consumption and chemicals are counted as indirect emissions. The carbon footprint (CF) of WWTPs is mainly from nitrous oxide (N_2O) fugitive emissions. In Finland, the share of direct emissions is 67%; from this, 52% is from nitrous oxide. The global warming potential (GWP) of N_2O is much greater than carbon dioxide, which explains its great share of total CF (Maktabifard *et al.* 2022).

Producing aeration energy (AE) for the nitrification process is one of the major energy sinks at WWTPs. AE consumption can vary between 54 and 70% of WWTP energy consumption (Gandiglio *et al.* 2017; Siatou *et al.* 2020). According to Crawford & Sandino (2010), in the USA, more than half of typical WWTP energy consumption is consumed by aeration. In Greece, it is 67.2% (Siatou *et al.* 2020).

This paper concerns the connections between food consumption and the nitrogen load to the WWTPs and their impacts on the environmental footprint of the treatment processes. The influent nitrogen load has an impact on AE consumption, methanol consumption for denitrification and the release of nitrous oxide from the active sludge treatment process. A more accurate understanding of the correlation between the protein content in food and the influent nitrogen load to the WWTPs allows for better predictions of future influent loads. This enables evaluating the feasibility of different treatment alternatives based on correct assumptions of the carbon-to-nitrogen ratio and the oxygen consumption pattern at WWTPs. Also, benefits from this assessment can indicate savings in WWTPs' energy and chemical consumption and a decrease in GHG emissions, which can also lead to economic benefits. When nutrient loads to water bodies decrease, less eutrophication occurs.

METHODS

Nitrogen load comparison between consumption and WWTPs

For the historical nitrogen load analysis, a comparison was made between the diet-related nitrogen load and wastewater treatment nitrogen loading. Food and protein consumption data as food supply and protein supply from Food Balance Sheets (FBS) (FAO 2001) were used. Food supply and protein supply data are available for the years 1961–2018 in Finland. Personal food and protein supply data are calculated for the whole country by dividing the available food and protein by the population (Sekula *et al.* 1991). Therefore, these values do not consider food waste or loss.

Nitrogen load to WWTP data is from the Finnish environmental institute (SYKE) (Säylä 2015; Lapinlampi 2021) for the years 1971–2013. These reports have the nitrogen load estimated for the centralized sewer systems in the whole country, meaning urban and semi-urban areas. According to Lapinlampi (2021), reporting has changed during these years. At the end of 2013, the population of Finland was 5.54 million and the share of urban and semi-urban inhabitants was 84.9% (4.57 million) (Säylä 2015).

The personal nitrogen load can be calculated by dividing the total nitrogen load by the number of urban and semi-urban inhabitants. In Finland, the urban and semi-urban wastewater nitrogen load per capita was around 14–16 g cap^{-1} d^{-1} between 1980 and 2010 (Säylä 2015). We estimated that the personal nitrogen load has not been presenting a realistic picture until 1995 due to the fact that the number of inhabitants within WWTPs rose rapidly in 1970–1980 (Figure 1), whereas the reporting of both inhabitants and influent loads was still unreliable.

Diet scenarios

Food and protein consumption are connected to dietary guidelines. For example, the Finnish Institute for Health and Welfare (THL) guidelines for protein intake in Finland is between 10 and 20% of the daily calorie intake, or 1.1–1.3 g kg^{-1} (Fogelholm *et al.* 2018). This varies somewhat between different age groups and people's activity levels (THL 2021). One gram of protein is assumed to contain 4 kcal (Atwater 1910).

Four scenarios were created to analyse the effects of food and protein consumption. The baseline of the dietary energy and protein supply was 3,343 kcal cap^{-1} d^{-1} and 118 g cap^{-1} d^{-1} (14%) (FAO 2001), respectively, in the year 2018. Food supply (kcal) means food available for consumption and food intake (kcal) consumed (not including consumption waste).

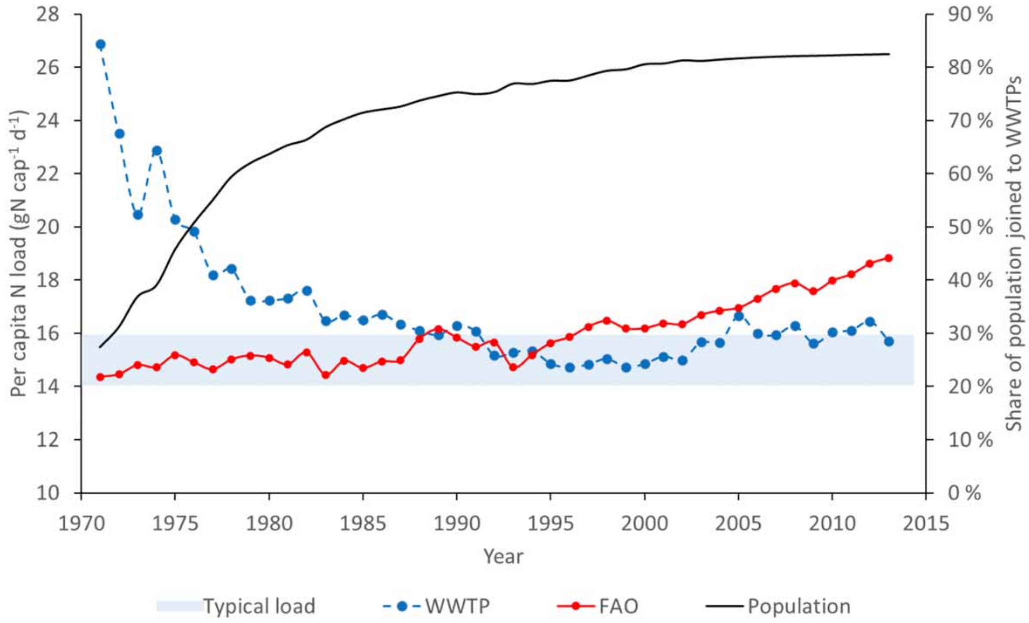

Figure 1 | Personal nitrogen load to the WWTPs and the food consumption between the years 1970 and 2013. WWTP, per capita N load from WWTP statistics; FAO per capita N load from FAO statistics; Typical load, typical per capita N load (Tchobanoglous *et al.* 2004); Population, the share of the population joined to WWTPs.

For estimating the nitrogen load from diets, FAO and THL recommendations were used. The *N decrease 1* scenario represents FAO's recommendation, called ADER – the average dietary energy requirement. This varies between countries. For Finland, the ADER was 2,548 kcal in 2018 (FAO 2001). Protein intake has been scaled to be 14% of the total energy supply which is the same as in 2018. *N decrease 2* represents the THL (Fogelholm *et al.* 2018) recommendation where the country's average energy requirement is 2,330 kcal d^{-1}. Protein intake is 15%, which is the average recommendation.

In addition, two cases where the share of protein has increased from the 2018 values are also examined, although the share of protein is kept within the recommended limit of 10–20%. For *N increase 1*, protein intake is 15%, and for *N increase 2*, protein intake is 20%. The total energy consumption in these scenarios is the same as the current FAO consumption, as it is hypothesized to be around 3,200–3,300 kcal in Europe in 2050 (FAO 2018).

Modelling nitrogen load impact at the plant

In this study, the Hermanninsaari WWTP in Porvoo, Finland (PE 55,000), was used as the reference for the process details and process performance. Phosphorus is removed chemically by adding ferrous sulphate before pre-aeration. Nitrogen is removed by an activated sludge process. More ferrous sulphate and polymers are added before secondary sedimentation. The typical Finnish WWTP uses a biochemical-activated sludge process that includes nitrogen removal (Laitinen *et al.* 2014).

This typical Finnish WWTP process was used to estimate the impact of the changes on the influent load. The model was built and simulated using Dynamita's SUMO process-simulating programme (v. 19.3.0) and the ASM3 biological model. The ASM3 model represents the denitrification–nitrification process well and also enables estimations of process energy consumption (Alex *et al.* 2018). Model default values were used. No detailed calibration was carried out, but we verified that the model predicted the nitrogen removal correctly. The data used in simulations were from 2021.

The first objective in the simulations was to keep the nitrogen removal performance the same as the plant achieved during the study period, and secondly, if the first objective could not be achieved, the plant's nitrogen removal requirement of 70% and nitrogen concentration below 15 mg/l as a yearly average were fulfilled. Nitrogen load exceeding the 2021 load was assumed to be removed by nitrification increasing the AE and by denitrification, requiring the addition of methanol as a carbon source in the process. The change in nitrogen load is modelled in maximum and minimum flows and temperature, which means spring and autumn flows in Nordic conditions. Spring is the period where the process operates on the edge of its capacity, and autumn represents the best performance of the WWTP in Nordic conditions.

The loading parameters of the simulation from two different loading conditions are shown in Table 1. The spring conditions are characterized by a high influent flow, low temperature and a sludge retention time (SRT) of 16.6 days. Autumn conditions are characterized by a normal influent flow, relatively high temperature and high SRT. The SRT for autumn was determined with the highest nitrogen load in order to have good performance in the treatment process. In the simulations, the sludge age was kept the same as it had been during the spring and autumn periods. In reality, the WWTPs might have sufficient capacity to increase the sludge age and increase the nitrifying biomass in the process. Changes in sludge age would have complex consequences on the AE need, sludge yield, settling and dewatering characteristics, and methane production potential. Nitrification slows down in lower temperatures, which can be compensated for by operating the process with a longer sludge age. These aspects of the process operation were omitted to simplify our simulations.

Calculating the environmental impact of N_2O, methanol consumption and AE

In this study, the environmental impact was evaluated based on the most influential emissions related to nitrogen removal. According to Maktabifard *et al.* (2022), direct emissions cause the highest impact, followed by energy consumption and chemicals. From direct emissions, nitrous oxide emissions were considered. From indirect emissions, methanol consumption and AE as carbon dioxide equivalents (CO_2-eq.) were considered. For calculating nitrous oxide, the emissions factor from IPCC (2019) is used. Using this factor, nitrous oxide emissions are directly proportional to the incoming nitrogen load, which is not accurate in real life. Nitrous oxide emissions have been shown to vary a lot between processes, and many other aspects besides the incoming nitrogen load would need to be considered. The widely used emission factor approach was adopted anyway since the recently updated emission factor reflects the results from several monitoring campaigns.

AE consumption calculations were done according to Alex *et al.* (2018):

$$\mathrm{AE} = \frac{S_O^{\mathrm{sat}}}{1.8 \times 1000}\left(\sum_{i=1}^{8} V_i K_L a_i\right)$$

where $S_{\mathrm{O}}^{\mathrm{sat}}$ is the dissolved oxygen concentration at saturation (8 g m^{-3}), $K_L a_i$ is the oxygen mass transfer coefficient in the reactors ($i = 1, \ldots, 8$) (1 d^{-1}) and $V_i =$ is the reactor volumes (m^3). The environmental impact of AE consumption depends on how the energy is produced and whether it is fossil-based or renewable (Maktabifard *et al.* 2022).

Only methanol was considered out of the many chemicals being used in the wastewater treatment process because its direct link to nitrogen removal serving as an external carbon source for denitrification. Consumption was only considered when the load to water bodies exceeded the results from 2021; the excessive nitrogen load is assumed to be removed only by using methanol. The assumption for methanol consumption is 3 mg of methanol per mg of nitrate nitrogen (Bajwa 2022). The values used in environmental impact calculations are presented in Table 2.

Table 1 | WWTP operation conditions of the simulations in spring and autumn

Variable	Unit	Spring	Autumn
Influent flow	m^3/d	13,972	5,382
Temperature	°C	9	15
Total chemical oxygen demand (TCOD)	mg/l	211	543
Total phosphorus (TP)	mg/l	3.2	7.5
SRT	days	16.6	20
Volumetric loading	kg BOD/m^3/d	0.090	0.063
Surface load in the secondary clarifier	m/h	0.32	0.12
Recycle ratio (return activated sludge + internal nitrate recycles)	%	96	250

Table 2 | Environmental parameters

Emission factor	Value	Unit
N_2O GWP	265	kg CO_2-eq./kg N_2O
N_2O (from wastewater treatment)	0.016	kg N_2O/kg $N_{influent}$
Methanol	462	g CO_2-eq./kg
Energy	0.24	kg CO_2-eq./kWh

RESULTS AND DISCUSSION

Historical data comparison

According to the inspection of the statistical information concerning WWTPs, the per capita nitrogen load decreased strongly from the year 1971 to the middle of the 1990s (Figure 1). At the same time, the share of the population that joined to WWTPs was rapidly increasing, but the reporting was not fully reliable. The per capita nitrogen loads to WWTPs in the 1970 and 1980 s were high compared to typical loads (Tchobanoglous *et al.* 2004). In this early historical data, these high loads were likely caused by errors in sampling and influent flow measurements at the WWTPs. For example, the internal recirculation from sludge dewatering was included in the influent water sample at many WWTPs. Furthermore, the flow rate measurement with a commonly used Venturi flume often exaggerated the flow. When each of the WWTPs improved its systems and water sampling locations, the influent loads were estimated more accurately. Moreover, the per capita nitrogen loads of five different Finnish WWTPs collected in our study from 2018 to 2020 were 14.3–21.7 gN/cap/d. These loads may have been slightly exaggerated because these WWTPs received industrial wastewater or septage from neighbouring municipalities. In addition, the highest of these loads was calculated for a WWTP receiving reject water from a waste digestion plant. The Finnish per capita nitrogen loads estimated in another report (Säylä 2015) were between 15 and 16 gN/cap/d for the years 1980, 1985 and 1990. Säylä (2015) utilized the same national data as our research; consequently, they excluded the non-essential loads with additional information. The details of their method are not available anymore. However, this emphasizes that the per capita loads calculated from the WWTP statistics for the years 1971–1990 (Figure 1) were exaggerated.

The per capita nitrogen load calculated from the FAO statistics increased almost linearly between the years 1971 and 2013. The values were plausible even from the beginning unlike the WWTP data. In our research, it was presumed that the per capita nitrogen loads calculated from FAO statistics would be higher than from the WWTP statistics because a share of the foodstuff would end up in the waste collection system (Antikainen 2007).

Thus, we compared only the statistical data between the years 1995 and 2013. This comparison arrived at a ratio of 0.9 for the per capita nitrogen loads calculated from the WWTP and FAO statistical data. The results from this period also confirmed the earlier observations by Van Drecht *et al.* (2009). The ratio was used in estimating the influent N load to the WWTP from dietary N consumption in the scenarios (see Table 3).

Impact of changing the diets

Comparing the results with the modelled current baseline, depending on the incoming nitrogen load, the nitrogen load to the treatment plant could decrease by 16–24% with FAO and THL recommended diets. If protein consumption slightly increases at the same time, the nitrogen load increases by 6%, and with a more significant increase, the nitrogen load could potentially increase by 42%. According to these results, an increase in protein consumption also increases the nitrogen load. This might also significantly increase the nitrogen load to receiving water bodies if no change to the required removal percentage is done. Nitrogen loads for all scenarios are presented in Table 3.

Spring and autumn results

The impact on the nitrogen removal at the plant is presented in Table 4. Removal percentages were kept at 70–71% for spring and 87% for autumn except for the scenario with the highest nitrogen load. The permit requirement for effluent nitrogen concentration is 15.0 g N m^{-3}, which was only exceeded in the highest N load scenario (16.6 g N m^{-3}). Environmental limits for nitrogen are checked as a yearly average; hence, the single transgression does not violate the permit requirements. Changes in

Table 3 | Scenario-specific loads and total N-loads

Scenario	Dietary N load (g N/cap/d)	Total dietary N load (kg N/d)	Personal N load to WWTP (g N/cap/d)	N load to WWTP (kg N/d)	Change in N load (%)
WWTP data	18.9	799	17.0	654	0
N decrease 1	14.4	609	12.9	499	−24
N decrease 2	15.8	670	14.2	549	−16
N increase 1	20.1	849	18.0	695	6
N increase 2	26.7	1,131	24.0	927	42

Table 4 | Results considering spring and autumn simulations

Scenario	Leaving load (kg N/d)	Removal (%)	Change compared with WWTP data (%)
Spring			
WWTP data	192	71	0
N decrease 1	145	71	−33
N decrease 2	161	71	−20
N increase 1	205	70	6
N increase 2	281	70	32
Autumn			
WWTP data	85	87	0
N decrease 1	65	87	−31
N decrease 2	72	87	−19
N increase 1	91	87	6
N increase 2	178	81	52

the removal process were similar in spring and autumn. Diet recommendations decreased the nitrogen load by 19–33% and future scenarios by 6–52%.

Environmental impact (CO_2-eq.) of N_2O, methanol consumption and aeration

N_2O played a major role in the total environmental impact in the context of this work. The significance of methanol was only raised in the scenarios where it was used. Otherwise, the proportion of N_2O was about 90%. Environmental impact as t CO_2-eq./ a and changes between scenarios are presented in Table 5. AE consumption in autumn was on average 15% higher than that in spring. Scenarios where nitrogen load decreased consumption were 6–11% less than in the modelled 2018, and scenarios with increased nitrogen load AE consumption were 2–14% higher.

Impact of the results

Anticipating human food consumption and how it will change is difficult. The connection between designing WWTPs and food consumption might be considered in the future. Countless variables affect food consumption from global and local perspectives. Observing the whole food system, consumption has a fractional impact on the environment compared to the production of food. Dietary change can be affected by income level, population growth and geographical location. The knowledge of the impacts on wastewater treatment can be used to inform consumers about this aspect.

Error sources

Using FBS average values can slightly affect the accuracy of the results, as traditional diets vary somewhat by region. WWTPs in Finland also receive some amount of industrial wastewater, which increases certain nutrient loads. In addition, nitrogen load and WWTP participant data are somewhat laboriously available, and the data can have some accuracy

Table 5 | Environmental impact of nitrous oxide, methanol consumption and AE consumption

Scenario	Change in N_2O load (%)	N_2O emissions (t CO_2-eq./a)	Methanol consumption (t/a)	Methanol emissions (t CO_2-eq./a)	Change in AE consumption (%)	AE consumption emission (t CO_2-eq./a)	Total environmental impact (t CO_2-eq./a)	Change (%)
Spring								
WWTP data	0	1,013	21	9,820	0.0	75	10,909	0
N decrease 1	−33	850	0	0	−10.8	71	920	−92
N decrease 2	−20	772	0	0	−6.9	68	840	−92
N increase 1	6	1,076	36	16,436	2.4	77	17,589	61
N increase 2	32	1,435	119	54,760	13.3	87	56,282	416
Autumn								
WWTP data	0	1,013	0	0	0.0	89	1,102	0
N decrease 1	−31	850	0	0	−5.7	84	934	−15
N decrease 2	−19	772	0	0	−9.1	82	854	−23
N increase 1	6	1,076	0	0	2.0	91	1,167	6
N increase 2	52	1,435	26	12,133	13.7	103	13,671	1,141

problems and lack reliable documentation. Documentation methods have varied between treatment plants. The most notable error source can be a model that is not precisely calibrated. Incoming wastewater is assumed to have the same chemical oxygen demand (COD) fractions as the default SUMO1 model. The model thus overestimates the amount of incoming biodegradable carbon (Sihvonen 2018), which results in lower methanol consumption than in reality. In general, nitrification models are well known and studied, which supports the reliability of the results (Alex *et al.* 2018).

CONCLUSIONS

This paper provides a technology perspective on diet recommendations and offers valuable information for utilities. Communicating this aspect to consumers might open up new opportunities to influence incoming loads.

Dietary nitrogen affects the incoming nitrogen load to the WWTP. As people consume more nitrogen, more of it will end up in wastewater. This paper studied the connection between protein consumption and nitrogen load coming to the WWTP and how it affected its processes. Our results support the conclusion that these two have a strong connection based on historical data and simulations. Decreasing protein consumption would also decrease the nitrogen load to WWTP and GHG emissions at plants. The recommended diets would decrease the nitrogen load by 16–24% and the environmental impact of the wastewater treatment by 15–92%. Consuming 6–42% more protein than current consumption would increase the nitrogen load by the same amount (6–42%) and the environmental impact of the wastewater treatment by 6–1141%. The main contributor to the environmental impact of WWTPs in this work was N_2O emissions. Depending on the scenario, N_2O emissions cover around 90% of all environmental impacts considered in this work. The use of methanol significantly increased the environmental impact of scenarios and, at the same time, lowered the proportion of N_2O emissions. The applicability of the results outside of Finland is limited to countries with similar food consumption habits and WWTP technologies.

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

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