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DOI
10.1016/j.ecolecon.2023.107815

Publication date
2023

Document Version
Final published version

Published in
Ecological Economics

Ecological Economics 208 (2023) 107815

Contents lists available at ScienceDirect

Ecological Economics

journal homepage: www.elsevier.com/locate/ecolecon

The distributional effects of a nitrogen tax: Evidence from Germany

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ARTICLE INFO

Keywords:
Nitrogen tax
Environmental policy
Distributional effects
Household impacts
Germany

ABSTRACT

The high level of nitrogen emissions over the last decades and their adverse impact on the natural environment and human health are a pressing environmental issue. A nitrogen tax can be a cost-efficient and effective policy instrument to reduce nitrogen emissions. However, adverse effects on low- and middle-income households might lead to societal and political frictions that could end up in resistance. In this paper we investigate how a hypothetical nitrogen tax covering the specific external costs of nitrogen could be implemented and estimate its short-term distributional effects on household income groups in Germany. The findings show that the proposed tax would be regressive. However, if the tax rate is set equal to the true cost of nitrogen, the monetary impacts would overall be small, ranging from 1.15% of income for the first income quintile to 0.66% for the fifth. Complementary policy measures to lower the burden on low-income households, farmers and the energy sector could preempt social resistance against the tax.

1. Introduction

Nitrogen (N) is a vital nutrient for plants, animals, and humans and has played an essential role in satisfying the increasing demand for food and energy during the last century (Erisman et al., 2013; Bashir et al., 2013). While the amount of reactive N (Nr) available for living organisms has long been limited by natural processes, humans have accelerated the conversion of unreactive N (N_2) in Nr compounds (Galloway et al., 2003). Global Nr emissions have risen ten-fold since the mid-19th century largely due to the combustion of fossil fuels and the introduction of the Haber-Bosch process, a method for directly synthesizing ammonia from hydrogen and nitrogen. It currently is the most economic process for the fixation of nitrogen (Britannica, 2023) and therefore the major source for ammonia, enabling the intensification of agricultural production systems (Galloway et al., 2003).

Current production at industrial scale utilizing the Haber-Bosch process requires high pressure and high temperature environments. Atmospheric nitrogen reacts with hydrogen, gained from methane (through steam reforming and water gas shift reaction) to ammonia (Klerke et al., 2008). The planetary boundary for the global N cycle might already be under severe pressure (Rockström et al. (2009) and Steffen et al. (2015)). Even worse, global Nr emissions are expected to more than double from 2010 to 2050 if no measures are taken (Bodirsky et al., 2014). Hence, an urgent effective solution is required.

1.1. Nitrogen emissions and impacts

Nitrogen has a versatile appearance. In contrast to the elementary form N_2, Nr is flexible and can take several forms. These include inorganic oxidized forms such as nitrogen oxide pollutants (NO_x), nitrous oxide (N_2O), nitrate (NO_3), inorganic reduced forms such as ammonia (NH_3) and ammonium (NH_4), and organically bound N (Norg) (Galloway et al., 2003). Agriculture and energy generation from both stationary and mobile combustion sources are responsible for a large part of these emissions (Hertel et al., 2012; Tian et al., 2020; Galloway et al., 2003; Jaeglé et al., 2005). Furthermore, industrial manufacturing processes also emit a considerable amount of Nr emissions, as the production of synthetic N-fertilizers releases atmospheric NH_3-emissions (Sutton et al., 2000).

The impacts of Nr emissions on the environment and human health are diverse. First, nitrous oxide is a highly potent greenhouse gas (IPCC, 2013). Second, several Nr emission compounds cause terrestrial and aquatic eutrophication, causing loss of biodiversity (Galloway et al., 2003). Thirdly, nitrogen dioxide (NO_2) and ammonia emissions have been associated with causing respiratory diseases (Brunekreef and Holgate, 2002). Being able to rapidly convert from one reactive form to

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https://doi.org/10.1016/j.ecolecon.2023.107815
Received 23 January 2022; Received in revised form 4 February 2023; Accepted 8 March 2023
Available online 23 March 2023

another, one single atom of Nr can cause manifold effects in different environmental systems. This phenomenon is called the N cascade (Galloway et al., 2003; Erisman et al., 2013).

1.2. An economic solution for the nitrogen problem

Nitrogen use can be viewed as a classical economic problem of an *environmental externality*. Actors optimize their individual welfare when using nitrogen without considering the impacts on third parties (Pigou, 1920). The use of nitrogen causes external costs, i.e. negative effects on ecosystems and other actors. Standard economic theory suggests that an environmental tax is a cost-efficient policy instrument for reducing harmful emissions (Fullerton et al., 2010). A Pigouvian tax has been prominently advocated for global anthropogenic climate change by economists (Weitzman, 2014; Cramton et al., 2017; Edenhofer et al., 2015; High-Level Commission on Carbon Prices, 2017), ideally covering the social cost of carbon. In a similar way an ideal nitrogen tax would internalize the external cost of nitrogen emissions, so that all actors (producers and consumers) face the true associated cost in their decision making and adjust their behaviour accordingly, exploiting all viable abatement options. The socially optimal level of emissions would result, maximizing the overall net benefits for the use of nitrogen.

1.3. Associated challenges

There are specific potential problems linked to the implementation of an environmental tax. Any tax reform will have distributional effects and create (relative) winners and losers – which will also hold true for nitrogen. Depending on the level of impacts, such a tax can trigger societal and political resistance due to negative short-term social impacts, particularly on poor or middle-class households (Vogt-Schilb and Hallegatte, 2017; Rentschler and Bazilian, 2017). The literature identified different sub-dimensions that need special attention such as dealing with hardship cases, job losses, single influential stakeholders, potential relevant price shocks for key commodities, reduced competitiveness of industries or poor levels of government or institutions that might act as catalyser (Rentschler and Bazilian, 2017; Carattini et al., 2018).

The social costs of nitrogen emissions are difficult to quantify. First, similar to the case of carbon emissions, parts of the social costs depend on discount rates, which are subject to ethical debates (Llavador et al. (2013)). Second, the diverse and versatile character of nitrogen means that depending on the current state, i.e. the current molecular appearance within the nitrogen cycle different underlying characteristics need to be considered that have an influence on the actual 'harm', making a precise quantification more difficult. In addition, to some extent the damages are location-specific because they depend on the characteristics of the ecosystem to which nitrogen is released.

Current estimates are scarce. Shindell (2015) assesses the social costs of atmospheric releases, covering impacts per ton of N_2O, NO_x and NH_3, using different discount rates and climate, climate-health and composition-health dimensions as impact categories. The results indicate that social costs of atmospheric releases of N_2O are in the range of 1–8\$ per ton, ~25\$ per ton of NO_x and ~ 10\$ per ton of NH_3 (in 2007 prices). For the European Union (EU) in 2008, the total social cost of nitrogen (total SNC), i.e. the monetized total environmental damages associated with Nr emissions, amounted to 75–485 G€ (Van Grinsven et al., 2013). This number implies a per-capita social costs of nitrogen of 152–980€ (in 2008 prices and reflecting the EU population of 495 million). If these costs were internalized through a tax, the additional expenditures for low income households could be substantial. The impact might be further skewed towards low- and middle-income households as they spend a higher share of their income on Nr emission-intensive basic goods such as energy and food (Liang et al., 2018; Feindt et al., 2021). An ex-ante assessment of the distributional impacts of a nitrogen tax has not been conducted, yet.

1.4. Related literature on distributional impacts

Several studies assessed the monetary impact on households by carbon pricing. Labandeira and Labeaga (1999) find a carbon tax to be weakly progressive in Spain. But most studies analyzing the incidence of a carbon or energy tax establish regressivity, in the US (Fremstad and Paul, 2019; Mathur and Morris, 2014), in Denmark (Klinge Jacobsen et al., 2003), in the UK (Feng et al., 2010), and in Germany (Hardadi et al., 2021). Recent cross-country studies have revealed country and income-level dependent patterns (Dorband et al., 2019; Feindt et al., 2021; Steckel et al., 2021).

Few studies include additional GHGs, among others N_2O, through which food products are taxed more heavily. Non-CO_2 emissions have been considered in carbon tax simulations, taking the global warming potentials as benchmark for establishing a CO_2-equivalent.[1] Among those, regressive effects are observed for Mexico (Renner et al., 2018) and Brazil (Da Silva Freitas et al., 2016), whereas outcomes are shown to be progressive for the Netherlands (Kerkhof et al., 2008). Studying Latin American and the Caribbean countries, Vogt-Schilb and Hallegatte (2017) find predominately regressive outcomes. Assessments of transport fuel taxes (nitrogen emissions are also released through combustion of fuels) are more mixed. Outcomes turn out to be regressive in the US (Teixidó and Verde, 2017; Chernick and Reschovsky, 1997), neutral in Germany (Flues and Thomas, 2015; Sterner, 2012) and progressive in Denmark (Klinge Jacobsen et al., 2003) and in Hungary (Flues and Thomas, 2015).

There are similarities between carbon and nitrogen taxes, but also differences. In addition to price increases of energy-intensive goods, prices for food can be expected to rise as well, since Nr emissions stem largely from the agricultural sector. A clear conclusion about the distributional effects of such a tax, however, cannot be drawn as there are no studies covering emissions of other N compounds, apart from N_2O.

1.5. Simulating a nitrogen tax in Germany

We fill this knowledge gap by assessing the distributional short-term impacts of a nitrogen tax in Germany. Considering that most of the German population disapproves the already high current degree of income and wealth inequality in Germany (Mau and Heuer, 2017), implementing such a tax would likely encounter resistance, if it turns out to be regressive. To increase social and political acceptability of a N tax, it is therefore crucial to understand the associated distributional effects beforehand and enable the design of potentially required compensation mechanisms (Klenert et al., 2018; Feng et al., 2018). In this respect, it is important to understand the net burdens for each single country and income group, as huge variation across countries exists (Fullerton and Metcalf, 2002; Fullerton, 2021).

We use a static IO approach established in the literature that consists of two main steps (see e.g. Steckel et al., 2021; Malerba et al., 2021; Feindt et al., 2021). First, the environmentally extended IO model simulates the price increase of sectoral goods and services due to the implementation of nitrogen taxation. The model assumes that price increases of goods and services are passed on entirely to final consumers and that producer and consumer demand functions are perfectly inelastic (see 3.7 for a more detailed discussion of this approach). Some Nr emissions such as from agriculture or waste management are difficult to observe or difficult to assign to a specific product. Hence, we either propose indirect measures (specifically an input tax on nitrogen fertilizer) or we exclude these emissions from the tax base. Moreover, NH4- and NO3-emissions are excluded, as the underlying IO database does not

[1] Global warming potentials can be different, depending on the considered time horizon and whether climate system feedbacks are considered, see IPCC (2021)

Table 1
Quantities and shares of important N-compounds and emitter groups in total anthropogenic Nr emissions in Germany in 2015.

Emitter group	NO_x [kt N/year]	NH_3 [kt N/year]	N_2O [kt N/year]	Sum [kt N/year]	Share
Agriculture	38.6	501.8	33.0	573.4	58%
Manure management	*0.4*	*210.8*	*3.3*	*214.6*	*22%*
Mineral fertilizer application	*21.0*	*56.5*	*8.6*	*86.1*	*9%*
Others	*17.1*	*234.5*	*21.1*	*272.7*	*28%*
Transport	191.4	9.2	1.6	202.2	21%
Energy and Industry	184.4	13.1	5.5	203.0	21%
Waste treatment	0.2	2.8	0.9	4.0	0%
Sum	414.6	526.9	41.0	982.6	
Share	42%	54%	4%		100%
Sum	1368.1 [kt NOx/year]	640.7 [kt NH_3/year]	129.0 [kt N_2O/year]		

Note: "Others" includes Nr emissions from crop residues from the application of sewage sludge and digested energy crops, as well as from animal grazing. Source: Umweltbundesamt (2020a), Umweltbundesamt (2021), Rösemann et al. (2021).

contain the respective satellite accounts (see 3.3 for further details).

Previous studies solely consider N_2O emissions and the associated equivalent carbon tax through global warming potential (GWP) (e.g. Kerkhof et al., 2008; Da Silva Freitas et al., 2016; Renner et al., 2018). Other impacts of the nitrogen cascade and other nitrogen emissions have not been covered. We assess multiple sources of N-emissions and consider their respective social costs. In case of N_2O we also apply characteristic specific social costs to set the level of the tax, instead of using GWP as a conversion factor. In a following step, we merge the price increases with household consumption data and calculate the additional consumption expenditures resulting from the price changes by household income quintile, assuming constant consumption patterns.

Section 2 describes the nitrogen problem in Germany. Section 3 presents the data and the methods used in this study. Section 4 reports the results: a nitrogen tax turns out to be regressive, but the additional expenditure burden tends to be rather small. Section 5 concludes with a policy discussion.

2. Excessive nitrogen use in Germany

Germany struggles with critically high Nr deposition levels. Due to excessive NO_3 concentrations about 27% of all groundwater bodies did not comply with the European Water Framework Directive's specification of good chemical status in 2009, and 48% of Germany's natural and semi-natural terrestrial ecosystems surpassed eutrophication limits (SRU, 2015). NO_2- as well as NH_3-emissions still exceed European air quality standards (Salomon et al., 2016; Umweltbundesamt, 2020b; Umweltbundesamt, 2021). German N_2O-emissions have decreased by 46% from 1990 to 2019 (Umweltbundesamt, 2020a), but atmospheric N_2O concentrations have not (Salomon et al., 2016). Table 1 presents an overview of the contributions of different emitter groups to NO_x-, NH_3-, and N_2O-emissions.

In response to the European Nitrates Directive (Nitrates Directive, 1991) and the Water Framework Directive (Water Framework Directive, 2000), Germany implemented the legally binding Fertilizer Ordinance (Düngeverordnung, 1996) in 1996 and the Fertilizer Law (Düngegesetz, 2009) in 2009. These regulate maximum quantities of fertilizer use and stipulate minimum distances between fertilization areas and surface water bodies. Supplementary national measures have also been introduced (Wiering et al., 2020; Salomon et al., 2016), such as subsidies for farmers who implement certain green agricultural practices and advisory services for farmers who aim to adopt greener agricultural practices (Wiering et al., 2020; Salomon et al., 2016). Yet to this day Germany fails to comply with the directives' limits on NO_3-emissions, which triggered an infringement process by the European Commission in 2013. The continued non-compliance makes a second infringement procedure likely (Kirschke et al., 2019).

Germany's Nr emissions from combustion are regulated through the Federal Emission Control Act (Bundes-Immissionsschutzgesetz, 1974), inter alia addressing limits specified by the EU (Industrial Emission Directive, 2010). Within the context of the National Emission Ceilings Directive (National Emission Ceilings Directive, 2001), Germany's commitments to reduce air pollution from NH_3 and NO_x are viewed as insufficient (Salomon et al., 2016). Regular non-compliance with NO_2-emission limits triggered an infringement process by the European Commission (European Commission, 2017). Germany's policies to address Nr emissions, largely voluntary measures, are widely viewed as insufficient for meeting reduction targets (Kirschke et al., 2019; SRU, 2015; Umweltbundesamt, 2014).

A nitrogen tax could be an efficient and cost effective way of dealing with the nitrogen issue. We simulate the immediate monetary impact of a nitrogen tax on consumption goods and assess the potential distributional outcome to understand societal conflict potentials.

3. Data and methods

The analysis draws on two main data sources. First, IO data, which is described in section 3.1, is used to calculate Nr emission and fertilizer intensities of products and respective price increases in Germany. Second, these price increases are combined with data on household income and expenditures, the latter being specified in section 3.2. Sections 3.3 and 3.4 explain the set-up of the tax mechanism regarding tax base and tax rates assumed. Section 3.5 addresses the method used for modelling the price increases induced by the hypothetical N tax. Finally, section 3.6 elaborates on the method used for analyzing the distributional effects of the price increases modelled beforehand.

3.1. Input-output table and emission data

The underlying data comprises a multi-regional IO table from Exiobase for the year 2015 (Stadler et al., 2021), which includes a national 200 × 200 commodity IO table for Germany. Its physical extension on Nr emissions covers N_2O-, NO_x-, NH_3- as well as Norg-emissions from combustion, non-combustion, agriculture, and waste processes.

3.2. Household income and expenditure data

We use Eurostat data on German household expenditures by income group (Eurostat, 2021b) for the year 2015. This dataset is based on the "Einkommens- und Verbrauchsstichprobe" (EVS) from the year 2013 and has been adjusted to the year 2015 by applying price coefficients (Eurostat, 2020). It includes 12 overall consumption categories and subcategories according to the "European Classification of Individual Consumption according to Purpose" (Eurostat, 2023). The dataset is based on a representative sample of around 52,400 German households, respectively around 10,480 German households for the expenditure categories food, beverages and tobacco (Eurostat, 2020).

Whenever possible in the merging procedure of IO data and household data, we use the specific consumption sub-categories available. Otherwise items are matched to the overall consumption category. We end up with 35 consumption categories (Appendix, Table C1 and A3). Eurostat provides expenditures by consumption category as a share of total expenditure.[2] Absolute expenditures by income quintile and category are hence calculated by multiplying the shares with the overall mean consumption expenditure of each income quintile (Eurostat, 2021a).[3] As Eurostat does not report on the related mean household net income per income quintile, we estimate these values conducting the same procedure as Eurostat does for providing the household data (2020, 2021a). We use information on the distribution of total net income among household deciles as provided by Statistisches Bundesamt (2013) for the EVS 2013 and adjust them to the year 2015 by using the consumer price index (Statistisches Bundesamt, 2021a).[4] Expenditures and income by income quintile can be found in Appendix A.

3.3. Tax base

Taxing emissions from diffuse sources is generally not practical since the emissions are either not observable or cannot be observed at a reasonable cost (O'Shea, 2002). Emissions from agriculture are a typical example for diffuse source pollution. Leaching and run-off of Nr applied as fertilizer is influenced by unknown variables like soil type and rainfall, which thereby makes it difficult to relate the resulting emissions to their source.

Consequently, we do not *directly* include Nr emissions from agriculture in the tax base. As Exiobase explicitly considers the sector *'N-fertilizer'* and bearing in mind that the agricultural sector is a main contributor to Nr emissions, we place an input tax on N-fertilizer, as an indirect approach to cover agriculture-related Nr emissions. Taxing inputs, i.e. factors generating pollution, rather than emissions, is a common solution to tackle diffuse source pollution (e.g. O'Shea, 2002).

Nr emissions from waste treatment are excluded as well, since the emissions from waste treatment cannot be easily assigned to a specific product, hence, to a responsible party. Moreover, NH_4- and NO_3-emissions are excluded from the tax base, since Exiobase does not provide data.

So far the resulting tax base comprises N_2O-, NO_x-, and NH_3-emissions from combustion activities, as well as NO_x- and NH_3-emissions from non-combustion industrial production processes, next to our N-fertilizer tax. These emissions relate to point sources since they occur at power or industrial plants.

We further address emissions resulting from household use, using specific items of the household expenditure database. Hence, household consumption of N-fertilizer, and of the most important energy carriers, namely gasoline, diesel, natural gas, and heating oil, are included in the tax base, as well. Due to limited data availability of NH_3-emission factors, only N_2O- and NO_x-emissions can be considered for the taxation of household energy consumption.

Finally, imports for final demand are assumed to be subject to a border tax adjustment (BTA). The objective of any BTA is to raise the price of imports by the same absolute amount as domestic products, thereby preventing competitive disadvantages for the domestic economy and emission leakage (Ismer and Neuhoff, 2007). For simplicity, we assume that the products from the different regions have the same price and nitrogen content regardless of where they are produced. This implies imports experience the same relative price increase as their domestic counterparts. This approach is justified as we focus on distributional effects that German households would experience, independently of where these products and services originate. To avoid competitive disadvantages for the German fertilizer industry, imports of N-fertilizer are also taxed in our impact analysis.

The resulting tax base covers around 96% of total NO_x-, 15% of total NH_3-, and 38% of total N_2O-emissions that occurred in Germany in 2015 according to Umweltbundesamt (2021) and Umweltbundesamt (2020a). The tax on N-fertilizer additionally addresses NO_3-emissions. Since the contribution of mineral fertilizer application to NO_3-emissions is not specified, the share of NO_3-emissions covered by the tax cannot be quantified.

3.4. Tax rates

According to Pigou (1920), a tax that aims to internalize negative externalities should be set equal to the marginal social costs. Estimating the social costs of nitrogen is especially challenging due to the complexity of the N cycle resulting from the different forms, fates, and effects of N (Nigon et al., 2019; Keeler et al., 2016). For that reason, some studies approximated the social cost of non-CO_2 GHGs by converting them to CO_2-equivalents based on their GWPs and then valuing these with the SCC (e.g. Kerkhof et al., 2008; Da Silva Freitas et al., 2016; Renner et al., 2018). Marten and Newbold (2012) show, however, that this shortcut can underestimate the social cost of N_2O by up to 24%. Therefore we use their N_2O-specific social costs. For the year 2015 and considering a 3% discount rate, Marten and Newbold (2012) estimated damage costs of $ 15,000/ton N_2O (in 2007 US$). We convert this value into 2015 €-equivalents using the consumer price index (U.S. Bureau of Labor Statistics, 2021) and the USD/EUR exchange rate for 2015 (Eurostat, 2021d), which results in a tax rate of 15.45€/kg N_2O. An approach based on the GWP would give a tax rate of approximately 7.35€/kg N_2O.[5]

Estimates of the social costs of nitrogen, especially of NH_3 and NO_x, are to a large extent location-specific. Impacts of these gases depend on, for example location-specific conditions like soil moisture (Hu et al., 2015) and soil pH (Baggs et al., 2010) of ecosystems. In the European Nitrogen Assessment, Brink et al. (2011) provide the first comprehensive analysis of the costs of N pollution in the EU for the year 2000, taking into account effects on climate, ecosystems, and human health. Van Grinsven et al. (2013) published an update of these costs for the year 2008 which also integrated climate cooling effects of NH_3 and NO_x and impacts on marine ecosystems. In this study we use the average values of the ranges estimated by Van Grinsven et al. (2013), which results in tax

[2] The expenditure share for "Water supply and miscellaneous services relating to the dwelling" is stated to be zero for all quintiles for the year 2015. Since this is not realistic, we assume expenditure shares for this sub-category to be the same as in 2005, which is the last year in which shares other than zero were reported. This assumption is justified by the observation that the mean expenditure share for "Water supply and miscellaneous services relating to the dwelling" within the European Union has almost remained the same from 2005 to 2015.

[3] Mean consumption expenditures are converted from Purchasing Power Standard to Euros by applying the Purchasing Power Parity for Germany for the year 2015 (Eurostat, 2021c).

[4] Imputed rentals for housing are deducted both from expenditures and income since they are a fictitious non-monetary expenditure category and income source and are thus not affected by the tax. Furthermore, some overall categories contain expenditures that are not included in their sub-categories. These expenditures are not taken into account when the sub-categories are used instead of the overall category, since they cannot be assigned to a specific sub-category. They are additionally deducted from income to create a comparable basis of income and total expenditures. The non-specified expenditures comprise 6.3% of the total expenditures of the first income quintile, 2.5% of total expenditures of the second quintile, 2.7% of total expenditures of the third income quintile, 2.8% of total expenditures of the fourth income quintile, and 3.5% of total expenditures of the fifth income quintile.

[5] The implied SCC is based on a meta-study by Wang et al. (2019), who reviewed current research about estimates of the SCC, and presents the average of the values that peer-reviewed studies estimated, considering a 3% discount rate. The GWP assumed for N_2O is 265 over a 100-year time horizon (IPCC, 2013).

Table 2
Tax rates used in this study.

Tax base	Tax rate
N_2O	15.45 €/kg N_2O
NO_x	11.30 €/kg NO_x
NH_3	25.70 €/kg NH_3
N-fertilizer	1€ / 1€ N-fertilizer[a]

[a] Assuming that the price of N-fertilizer is entirely determined by its Nr-content, a 100% price increase of the Nr-content in fertilizer implies a tax rate of 1€ / 1€ N-fertilizer.

rates of 25.70€/kg NH_3 and 11.30€/kg NO_x.[6]

Brink et al. (2011) estimated the social costs for fertilizer application in the EU for the year 2000 from 0.40€ to 6.80€/kg Calcium ammonium nitrate (CAN)-Nr, stating, that the upper boundary is just indicative. CAN, which has a Nr-content of 27%, is the most applied N-fertilizer in Germany (Meyer-Aurich et al., 2020). A range of studies modelling taxes on N-fertilizer for Denmark (Schou et al., 2000; Berntsen et al., 2003), Spain (Martínez and Albiac, 2006), and France (Jayet and Petsakos, 2013; Bourgeois et al., 2014) assume a 100% increase of the price of the Nr-content in chemical fertilizer. To do a robustness check, we assessed the price of Nr in CAN-fertilizer in Germany in the year 2015 to be 0.74€/kg CAN-Nr[7] (LEL Schwäbisch Gmünd, 2016). Hence, a corresponding N-fertilizer tax reflecting a 100% price increase for the Nr content would imply a doubling of the price, resulting in a tax on the Nr content of fertilizer of 0.74€/kg Nr. This value lies within the range from Brink et al. (2011) and will be used in this study as the tax rate on N-fertilizer. An overview of the tax rates used in this study is given in Table 2. The corresponding calculations are documented in Appendix A.

Considering that the social costs of NH_3, NO_x, and N-fertilizer are based on estimates for the year 2008, respectively 2000. However, as a more recent assessment for the European context is lacking, the modelled tax rates are based on these values.

3.5. Modelling of price increases

The first part of this section explains the method used to model price increases of products induced by the tax on Nr emissions, while the last section describes the slightly different steps taken to model price increases of products caused by the tax on N-fertilizer. Furthermore, it elaborates on how the price increases of products are translated into price increases of expenditure categories.

3.6. Nr emissions taxation

In our analytical framework, Nr emissions can stem from three different sources. First, Nr emissions that occur from energy consumption directly at the household constitute the household emission component (N^{HH}). Second, the use of energy and of chemical products that lead to direct Nr-emissions during the production of a particular good represent the direct emission component (N^{dir}). Third, the emissions occurring during the production of other goods that are used as input for the production of a particular good account for the indirect emissions (N^{ind}). The latter requires trancing all upstream production steps in supply chains.

We define direct and indirect emissions as production emissions (N^P). Total emissions N^T (kg N-compound) are the sum of production and household emissions:

$$\underset{(ne\times np)}{N^T} = \underset{(ne\times np)}{N^P} + \underset{(ne\times np)}{N^{HH}} \tag{1}$$

where the dimension of N^T, N^P and N^{HH} is the number of Nr emission compounds ne by the number of products np in Exiobase.[8]

To quantify the price increase of goods consumed by households depending on the embodied Nr emissions, we use sectoral, respectively product, emission intensities, which is a frequently used approach in the literature (see e.g. Steckel et al., 2021; Feindt et al., 2021; Malerba et al., 2021). We compute household Nr emission intensity NI^{HH} (kg N-compound/€ energy product) of energy products consumed directly by households based on physical emission factors and price data per unit energy product. Calculation of production Nr embodied emission intensity NI^P (kg N-compound/€ product) is based on an IO framework (for an overview see Miller and Blair, 2009) as follows:

$$\underset{(ne\times(np^*nr))}{NI^P} = \underset{(ne\times(np^*nr))}{NI^{dir}} \underset{((np^*nr)\times(np^*nr))}{(I-A)^{-1}} \tag{2}$$

where NI^{dir} is the direct Nr-emission intensity (kg N-compound/€ product), $(I\text{-}A)^{-1}$ the Leontief inverse L and nr represents the number of regions in Exiobase. NI^{dir} is determined as:

$$\underset{(ne\times(np^*nr))}{NI^{dir}} = \underset{(ne\times(np^*nr))}{N^{dir}} \underset{((np^*nr)\times(np^*nr))}{x^{-1}} \tag{3}$$

where x represents total output by product and region. Since the tax base of the N tax are Nr emissions occurring in Germany, Nr emissions from foreign sectors in N^{dir} are set to zero.[9] Furthermore, as the focus of this analysis is on the German products, the dimension of NI^P is reduced to (ne × np) for the following calculations, where np represents the number of German products in Exiobase.

Total Nr embodied emission intensity NI^T (kg N-compound/€ product) is simply the sum of embodied production and household Nr emission intensity:

$$\underset{(ne\times np)}{NI^T} = \underset{(ne\times np)}{NI^P} + \underset{(ne\times np)}{NI^{HH}} \tag{4}$$

Finally, the relative price increase of product j due to the introduction of the tax on embodied emissions of N-compound e is determined by:

$$\frac{p_{j,e}^1 - p_j^0}{p_j^0} = t_e{}^*ni_{j,e}^T \tag{5}$$

where $p_{j,e}^1$ stands for the price of product j after the tax on embodied emissions of N-compound e has been implemented, p_j^0 for the price of product j before the implementation of the tax, and hence $\frac{p_{j,e}^1 - p_j^0}{p_j^0}$ for the relative price increase of product j due to the tax, which is calculated by the right-hand side of the equation. The determinants of the relative price increase are t_e, which represents the tax rate on embodied emissions of N-compound e (€ tax/kg N-compound), and $ni_{j,e}^T$, the total

[6] We convert the social costs by Van Grinsven et al. (2013) from €/kg Nr into €/kg NH_3, respectively €/kg NO_x, by using the respective molar mass ratios. For NO_x, which includes nitric oxide (NO) and NO_2, we assume the molar mass of NO_2 because although more NO gets emitted, it oxidizes to NO_2 relatively quickly afterwards (Umweltbundesamt, 2020c). Furthermore, the values are adjusted for inflation into 2015 €-equivalents using the consumer price index (Statistisches Bundesamt , 2021a) and taking 2000 as the base year. The by Van Grinsven et al. (2013) estimated ranges are based on values from 1995 to 2005, but since the original sources are not accessible, the exact year of each value cannot be identified.

[7] We manually searched for CAN fertilizer prices, which is independent of the used IO and household data. In a second step we converted the price given by LEL Schwäbisch Gmünd (2016) for the year 2016 into 2015 prices by using the consumer price index (Statistisches Bundesamt, 2021a).

[8] The following notation is used throughout this study: matrices are displayed in upper-case, vectors in lower-case, and scalars in italicized letters. Row vectors additionally carry the superscript ′.

[9] Finally imports are treated as having the same nitrogen emissions as locally produced goods. This approach is similar to the analysis of Malerba et al., 2021.

emission intensity of product j regarding N-compound e (kg N-compound/€ product). The unit of the result of eq. (5) is thus € tax/€ product, respectively %.

3.7. N-fertilizer taxation

The simulation of the N fertilizer taxation is done analogously to the Nr tax simulation. Like Nr emissions, the use of N-fertilizer can be divided into a household (f^{HH}) and a production (f^{P}) component. f^{HH} represents the N-fertilizer demand of households, while f^{P} is the N-fertilizer consumed during the production process of goods. Total N-fertilizer consumption f^{T}(€) is the sum of production and household consumption:

$$\underset{(np\times1)}{f^{T}} = \underset{(np\times1)}{f^{P}} + \underset{(np\times1)}{f^{HH}} \tag{6}$$

The tax base for taxation of demand of N-fertilizer by households is simply the amount of N-fertilizer consumed by households measured in € (f^{HH}), so no intensities are required. Price increases of products due to embodied N-fertilizer are quantified based on so-called production N-fertilizer intensities fi^{P} (€ N-fertilizer/€ product). The concept of N-fertilizer intensities is the same as Nr (embodied) emission intensities and their calculation is - analogue to Nr emission intensities - based on an IO framework. However, instead of using the physical extension of Exiobase, a simplified version of the price-shifting model by Coady (2006) as in Schaffitzel et al. (2020) is applied. Price changes of N-fertilizer are transferred to other products according to the elements of the Leontief inverse L. The single elements of the Leontief inverse (l_{ij}) represent the entire (direct and indirect) inputs of product i that are required to produce one unit of product j (Miller and Blair, 2009). Hence, l_{ij} is the monetary value of inputs from product i required to produce one € of product j. For instance, if the Leontief element for N-fertilizer input into wheat production was 0.2, this would imply that along the entire supply chain, 0.20 € of N-fertilizer are required to produce 1.00€ of wheat. Thus, if the price of N-fertilizer doubled - ceteris paribus - assuming that cost increases for producers are completely passed on to consumers, the price of one unit of wheat would rise by 20%.

Following the abovementioned example, fi_j^P of product j is determined by aggregating the elements of the Leontief matrix as follows:

$$fi_j^P = \sum_r l_{ij,r} \tag{7}$$

where i represents N-fertilizer and r stands for the regions that provide the N-fertilizer.

Total N-fertilizer embodied intensity fi^{T} is the sum of production N-fertilizer embodied intensity and household demand intensity of N-fertilizer fi^{HH},[10] which is the fertilizer consumption divided by the spent amount of money:

$$\underset{(np\times1)}{fi^{T}} = \underset{(np\times1)}{fi^{P}} + \underset{(np\times1)}{fi^{HH}} \tag{8}$$

The relative price increase of product j due to the introduction of the N-fertilizer tax can then be analogously calculated as follows:

$$\frac{p_{j,f}^1 - p_j^0}{p_j^0} = t_f * fi_j^T \tag{9}$$

where $p^{1}_{j,f}$ stands for the price of product j after the tax on N-fertilizer has been implemented, t_f for the tax rate on N-fertilizer (€/€ N-fertilizer), and fi_j^T for the total N-fertilizer intensity of product j.

[10] Implicitly fi^{HH} is zero for all elements other than fertilizer.

3.8. Overall price increase

Total relative price increase of product j[11] is the sum of its relative price increase due to the N-fertilizer tax and of its relative price increase due to the Nr emission taxes of each N-component *e*:

$$\frac{p_j^1 - p_j^0}{p_j^0} = \frac{p_{j,f}^1 - p_j^0}{p_j^0} + \sum_e \frac{p_{j,e}^1 - p_j^0}{p_j^0} \tag{10}$$

The relative price increase of expenditure category c is determined by means of a bridge matrix as follows:

$$\frac{p_c^1 - p_c^0}{p_c^0} = \underset{(1\times np)}{b'_c}\underset{(np\times1)}{pi_p} \tag{11}$$

where p_c^1 represents the price of expenditure category c after the tax has been implemented, p_c^0 the price of expenditure category c before the implementation of the tax, b_c is the row of the bridge matrix with expenditure category c and pi_p is a vector that contains the price increases by product as calculated in Eq. (10). We use the bridge matrix from Wood (2021) and apply changes where deemed appropriate, for instance for the German electricity mix (details in Appendix A).

3.9. Analysis of distributional effects

Additional expenditures (ae, in €) of income quintile q to maintain current consumption are calculated as follows:

$$ae_q = \sum_c \frac{p_c^1 - p_c^0}{p_c^0} * ce_{c,q} \tag{12}$$

where $ce_{c,\,q}$ represents the current mean expenditures of income quintile q for expenditure category c (€).

Monetary impacts (mi, in %)[12] for income quintile q are then calculated as follows:

$$mi_q = \frac{ae_q}{inc_q} \tag{13}$$

where inc_q is mean household income for income quintile q (€). This reflects the additional spending by households necessary to maintain their consumption, if no adjustments, such as substitution or production structure changes, would take place. Throughout this paper, monetary impacts refers to this definition.

Finally, the distributional effects of the tax result from the comparison of monetary impacts of the different household income quintiles.

[11] Whereas the calculated price increases due to production emission and fertilizer intensities refer to producer prices, the price increases due to household emission and fertilizer intensities as well as the data on household expenditures is given in purchaser prices. Exiobase, however, does not provide product-specific trade and transport margins, which would be needed for a conversion. Therefore, the difference between the price concepts that arises due to the trade and transport margins is not considered in this study.

[12] Poterba (1989) argues that household well-being is better approximated by annual expenditures than by annual income. He reasons that annual income can fluctuate considerably during a person's lifetime, and that annual expenditures, which are more constant over long periods of time, might provide a more reliable indicator of household welfare. For comparison, we also provide monetary impacts normalized by annual current expenditures in Figure C1 of the Appendix. Strictly speaking, for calculating monetary impacts normalized by expenditures by quintile, households would have to be assigned to quintiles based on their annual expenditures, not on their annual income. However, since we only have access to the aggregated data, not the microdata, we use the provided income quintiles. This implies the assumption that in either case – based on their annual income or their annual expenditures - households get assigned to the same quintile.

Fig. 1. Indirect and direct monetary impacts due to the introduction of a N tax by income quintiles (in %). Source: own calculations based on Stadler et al. (2021), Wood (2021), Eurostat (2021b), Eurostat (2021a), Statistisches Bundesamt (2013).

3.10. The static approach

In the literature similar static approaches have been applied to assess the impacts of carbon pricing (see e.g. Dorband et al., 2019; Feindt et al., 2021; Steckel et al., 2021; Malerba et al., 2021) and the impacts resulting from removing fossil fuel subsidies (Schaffitzel et al., 2020). The static approach estimates short-term impacts without considering behavioral changes and general equilibrium effects (Steckel et al., 2021; Malerba et al., 2021; Vogt-Schilb and Hallegatte, 2017). It disregards the price responses, as consumers and producers substitute products and change consumption patterns. Hence the static approach yields upper-bound estimates of the impacts. However, as argued by the scholars above, adjusting the consumption activities requires time. Especially poorer households have limited adaptation capacities in the short-run. Compensation schemes take time to work (Malerba et al., 2021; Fullerton, 2021), while the impacts are immediately felt, potentially causing political turmoil and social conflict. The static analysis therefore offers valuable insights for policy makers and involved actors. It can be expected that price changes, substitution effects, and behavioral adaptations contribute to an overall reduced impact (Dorband et al., 2019).

An alternative computable general equilibrium (CGE) approach could take relevant adjustment mechanisms into account in the assessment of long-term changes in production, consumption and welfare. Although superior in this respect, CGE models have difficulties with depicting supply chains in high detail and are not suitable for capturing short term effects and adaptation processes while transitioning from the original state to the new equilibrium (Fullerton and Muehlegger, 2019; Ward et al., 2019). Ideally CGE assessments and static IO approaches would be considered in parallel to provide policy makers with the best available information. Such a dual approach is however out of the scope of this study.

4. Results

Section 4.1 presents the results of our analysis. Section 4.2 reports more detailed on the tax-induced price increases of individual consumption categories.

4.1. Distributional effects

The overall monetary impacts on German households would be moderate. The monetary impacts, meaning the additional household expenditures, range from 0.66% of household income for the fifth income quintile to 1.15% for the first (Fig. 1). The outcomes show a clear and unambiguous regressive pattern, with continuous decline of the relative impact when income rises. Fig. 1 further distinguishes between direct and indirect effects. It shows that indirect effects, i.e. effects caused by taxation of production activities in supply chains, play a larger role than direct effects, which are responsible for ~1/6 of the total impacts. The relative importance of direct effects slightly increases with income and is the highest for the fifth income quintile, making up around 21% of its total monetary impact.

Decomposing the monetary impact by income quintile into the contributions of different Nr emissions and N-fertilizer shows that the tax on NO_x-emissions is responsible for the lion's share (see Fig. 2), where industries' NOx contributions are on average 4–6 times as large as households contributions. While individual taxation of N_2O-, NH_3-emissions or N-fertilizer leads to monetary impacts between 0.01 and 0.07%, a tax on NO_x-emissions creates monetary impacts between 0.59 and 1.04%. The relative contribution of N_2O- and NO_x-emissions to the overall monetary impact decreases with increasing income, whereas the opposite is observed for NH_3-emissions and N-fertilizer.

Richer quintiles spend absolutely more in almost every category than their lower-income counterparts. Regardless of the income quintile, the highest expenditure increases generally occur for the categories "Electricity, gas and other fuels", "Operation of personal transport equipment", "Other recreational items and equipment, gardens and pets", "Food", and "Transport services". The expenditure increases for

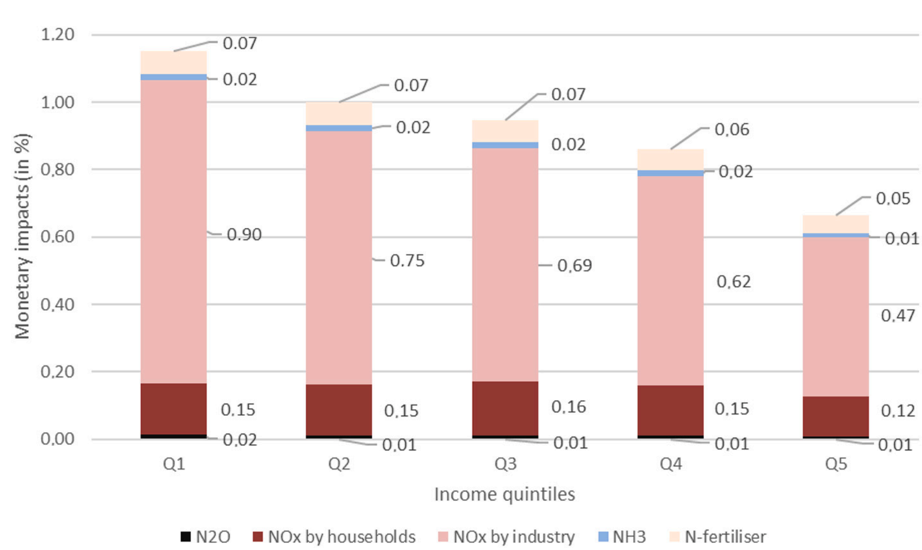

Fig. 2. Monetary impacts due to the introduction of a N tax by tax base and income quintiles (in %). Source: own calculations based on Stadler et al. (2021), Wood (2021), Eurostat (2021b), Eurostat (2021a), Statistisches Bundesamt (2013).

Table 3
Relative expenditure shares in % (rounded) of income quintiles of different product and service categories. Items where at least one entry exceeds 5% are highlighted.

	Q1	Q2	Q3	Q4	Q5
Total expenditure in €	11,937	18,07	24,210	31,209	45,304
Expenditure shares of income quintiles in %					
Food	15	13	13	13	11
Non-alcoholic beverages	2	1	1	1	1
Alcoholic beverages	1	1	1	1	1
Tobacco	1	1	1	1	0
Clothing	3	4	4	4	5
Footwear	1	1	1	1	1
Actual rentals for housing	28	22	14	9	4
Maintenance and repair of the dwelling	0	1	1	1	1
Water supply and miscellaneous services relating to the dwelling	1	1	2	3	3
Electricity, gas and other fuels	9	8	8	7	6
Furniture and furnishings, carpets and other floor coverings	1	2	2	2	3
Household textiles	0	0	0	0	0
Household appliances	1	1	1	1	1
Glassware, tableware and household utensils	0	0	0	0	0
Tools and equipment for house and garden	0	0	1	1	1
Goods and services for routine household maintenance	1	1	1	1	1
Medical products, appliances and equipment	2	2	2	2	2
Out-patient services	1	1	1	2	3
Hospital services	0	0	0	0	1
Purchase of vehicles	2	3	4	6	9
Operation of personal transport equipment	4	5	6	7	7
Transport services	2	2	2	2	2
Communications	4	4	3	3	2
Audio-visual, photographic and information processing equipment	1	1	1	1	2
Other major durables for recreation and culture	0	0	0	0	0
Other recreational items and equipment, gardens and pets	2	2	2	2	2
Recreational and cultural services	3	4	4	4	4
Newspapers, books and stationery	2	2	2	2	2
Package holidays	1	2	3	3	4
Education	1	1	1	1	1
Restaurants and hotels	4	5	6	6	7
Personal care	3	3	3	3	2
Personal effects n.e.c.	0	1	1	1	1
Insurance	3	5	5	6	6
Other financial services n.e.c.	0	0	0	0	0

"Electricity, gas and other fuels" and "Operation of personal transport equipment" are driven both by their high price increase as well as by the relatively high current expenditures for these categories (see Table 3 and Fig. 3). The expenditure increases for "Other recreational items and equipment, gardens and pets" as well as "Transport services", in contrast, can mainly be ascribed to their high price increase. Lastly, additional expenditures for "Food" can be attributed to the high current expenditures, which make up one of the largest shares of total expenditures, independent of income.

Adding up absolute expenditure increases (the relative price increase multiplied by the total expenditures) by income quintile multiplied by the number of households in each quintile, gives a total N tax revenue of around 11,061 M€ per year. This corresponds to about 7% of tax revenues generated by the value added tax in Germany in 2015 (Statistisches Bundesamt, 2021b). Around 90% (9902 M€) of total tax revenues can be attributed to the NO_x-tax, 7% (804 M€) to the N-fertilizer tax, 2% (218 M€) to the NH_3-tax, and 1% (136 M€) to the N_2O-tax.

Assessing monetary impacts by category and income quintile (see Table C2 of the Appendix) reveals the major drivers for the regressive effect. We find that "Electricity, gas and other fuels" is the main driver for the observed regressivity. While the relative monetary impact of this category is highest for the first income quintile, about 0.48%, it decreases steadily with income and comprises only about 0.19% for the fifth quintile. Low-income households tend to spend a higher share of their income for electricity and other energy products than high-income households. Another driver of regressivity is "Actual rentals for housing". For this category, the first income quintile experiences a monetary impact of 0.10%, whereas it amounts to only 0.01% for the fifth income quintile. The main intuition is that lower-income households tend to spend a larger share of their income on rent, whereas wealthier households tend to own the dwellings they live in.

4.2. *Price increases*

The tax-induced relative price increases are presented in Fig. 3.[13] Significant increases are observed for "Electricity, gas, and other fuels" (4.65%), followed by "Other recreational items and equipment, gardens and pets" (4.23%), "Transport services" (3.14%) and "Operation of personal transport equipment" (2.98%). Those four categories and "Water supply and miscellaneous services relating to the dwelling" (1.07%) see price increase above 1% . In contrast, the prices of the categories "Food", "Non-alcoholic beverages" (0.59%), and "Alcoholic beverages" (0.62%) increase only slightly. The reason is that N-fertilizer input only makes up for a small part of the value of food and beverages, which in turn is determined by a variety of other inputs like machinery and labour, as well as by depreciation and net operating surplus (see IO data used in this study). Service-oriented expenditure categories like "Restaurants and hotels", "Education", "Out-patient services", and "Hospital services" belong to the least affected categories, with price increases between 0.13 and 0.14%.

Fig. 3 also shows that except for "Other recreational items and equipment, gardens and pets", where a price increase is dominated by the N-fertilizer, price increases are driven almost solely by NOx-emissions. Further detail is given in Table C5 of the Appendix. Intensities by product can again be found in Table C4, please see Appendix B for further details on the underlying data and some minor data treatment.

Table A6 of the Appendix A shows which products are the main drivers of the price increase of individual categories. For instance, the price increase of "Electricity, gas and other fuels" is driven by a variety of energy products, with electricity by coal being the single most influential factor. The price increase of "Other recreational items and equipment, gardens and pets" is mostly determined by N-fertilizer, a product that is characterized by fairly high NH_3 and N-fertilizer intensities compared to the other products and is responsible for about 88% of the category's price increase.

5. Summary and concluding remarks

This study provides insights into the short-term distributional impacts of a nitrogen tax in Germany. It is the first such study that explicitly focuses on nitrogen, adding a new perspective to the literature environmental tax incidence, which has largely focused on carbon and energy taxes. Existing studies included N_2O in their analysis of the distributional effects of a GHG tax, and approximated the external costs based on the GWP and the SCC (e.g. Kerkhof et al., 2008; Da Silva Freitas et al., 2016; Renner et al., 2018). This study approximates the N_2O

[13] See Table C3 and C4, in the Appendix C for further elaboration

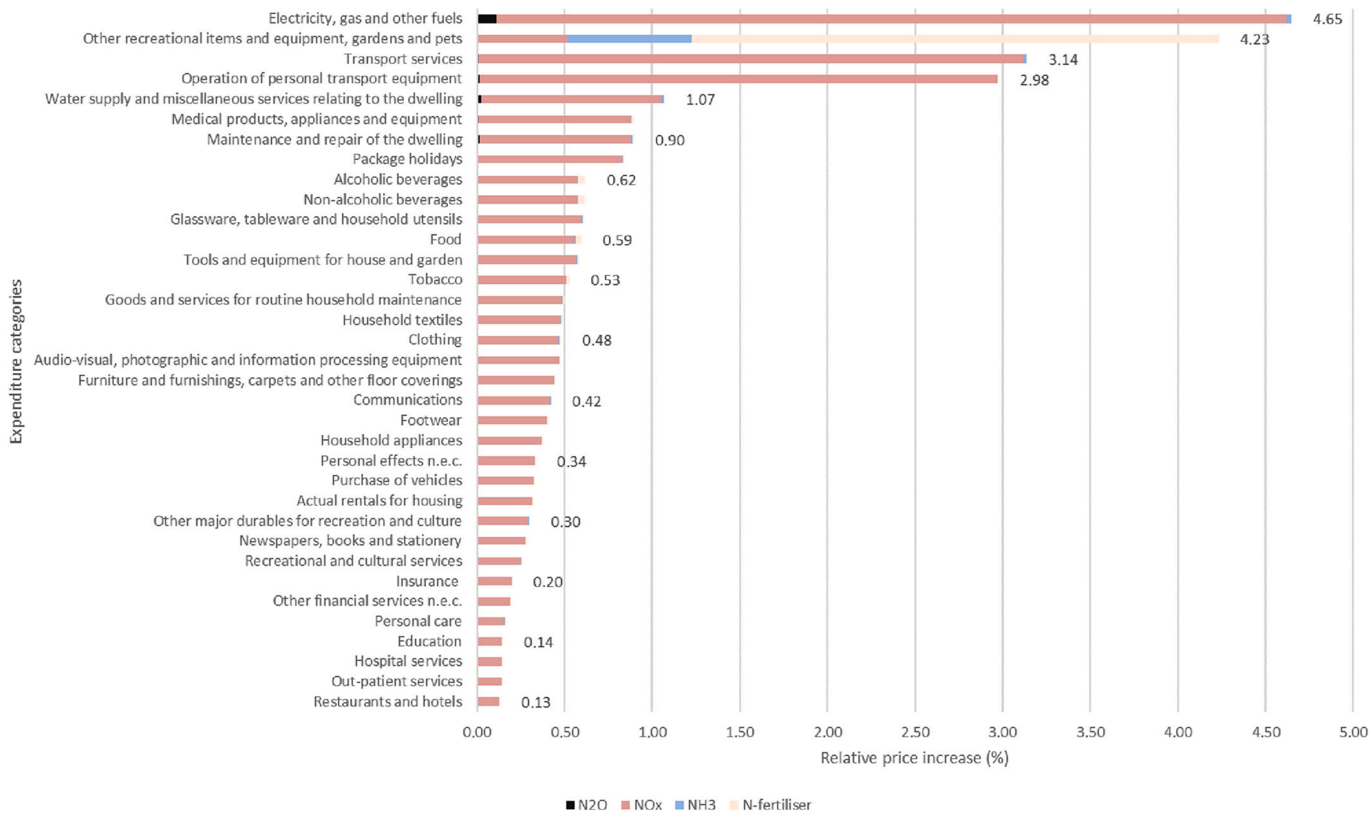

Fig. 3. Relative price increase (in %) due to the introduction of a N tax by expenditure categories and tax base. Source: own calculations based on Stadler et al. (2021), Wood (2021).

external damages by N_2O-specific costs. We provide policymakers an information basis for decisions regarding the design of the N tax policy, and highlight the need for compensation mechanisms to accompany environmental taxes.

Our study assesses the distributional impacts of a hypothetical N tax in Germany on different household income groups. We combine IO modelling to determine the tax-induced price increase of different goods and services with household expenditure data by product category to estimate resulting additional expenditure burden. The results show that a nitrogen tax would be regressive. The monetary impacts are relatively small, ranging from 0.66%–1.15% of household income for different income quintiles. Nevertheless, some product categories would see significant price increases and especially low-income households might perceive the hike in energy prices as a substantial negative impact on their budget. The largest price increases can be expected for “Electricity, gas and other fuels”, “Transport services” and “Operation of personal transport equipment”.

To reduce the adjustment costs and enhance the acceptance of the tax, the initial tax rates should be low and then subsequently increased. This might imply that a feasible approach cannot start with taxing the total social costs, but first has to start with a socially accepted taxation or burden level (Carattini et al., 2018). To avoid potential societal and political resistance and ensure inclusiveness, compensation measures could be designed for low-income households: lump-sum transfers, regular cash transfers, or vouchers, as has been suggested in other studies (e.g. Schaffitzel et al., 2020; Renner et al., 2018; Beck et al., 2015; Malerba et al., 2021). In addition, a clear, inclusive, convincing and transparent communication strategy is needed to foster acceptance (Rentschler and Bazilian, 2017) and avoid unintended snapbacks (Schaffitzel et al., 2020).

Likewise, compensation options should be explored to prevent resistance from farmers due to the considerable rise in the price of N-fertilizer and from stakeholders in the significantly affected energy sector. Compensation mechanisms can be financed by the tax revenues. Our estimates show that these account for around 11,061 M€ in the first year for a tax on NO_x-, N_2O-, and NH_3-emissions as well as N-fertilizer, and to around 10,707 M€ for a reduced tax base comprising only NO_x-emissions and N-fertilizer. Nevertheless, tax revenues can be expected to decline over the years due to adjustments of actors. Overall, we consider the implementation of a N tax viable under the aforementioned conditions. However, the implementation of a supportive BTA that prevents N leakage and competitive disadvantages for the domestic economy might be challenging.

It is generally questionable whether a BTA policy could be imposed in Germany since ensuring compliance with World Trade Organization rules can be difficult (De Cendra, 2006). Investigating possible ways to make a BTA policy work in practice is therefore crucial to increase feasibility of a N tax. Another option to reduce N leakage and competitive disadvantages for the domestic economy would be to directly implement an EU wide tax. This would additionally reduce Nr emissions in other member states, most of which face a similar N issue as Germany (Van Grinsven et al., 2013). The lessons from the debate about the practical and legal challenges of a carbon BTA, and how to overcome them, could inform the design of a nitrogen BTA (see e.g. Schmidt et al., 2021).

A first implementation of the N tax might be more practical if the initial tax base covered only NO_x emissions, since the administrative burden and monitoring costs would be lower. NO_x-emissions cover the lion’s share of total Nr-emissions. Considering that monitoring can be costly for smaller combustion sources, the tax could first be applied to large industrial plants.

Some methodological and measurement issues should be highlighted. First, the EVS household expenditure survey only includes households with a monthly net income below or equal to 18 K€ (Statistisches Bundesamt, 2013). This makes it difficult to provide a complete picture regarding the distributional effects of the tax, but it certainly allows to identify a tendency. Second, our analysis considers only immediate effects and neglects dynamic adjustment processes, notably substitution effects by industries and households. The burden on households and industries can be expected to be lower if they can switch to less Nr intensive goods. At the same time, if transport fuels become more expensive, wealthier households are more easily able to switch from an internal combustion engine vehicle to a battery electric vehicle. This tends to increase the regressive effect of the tax policy. First-order effects also do not reflect the effect of the tax on household income. Repercussions on income would likely reduce the policy's regressivity, as is the case for carbon taxes (Beck et al., 2015; Goulder et al., 2019; Sajeewani et al., 2015). However, first-order effects are reasonable upper-bound estimates and give insight into the short-term distributional impacts of the tax (Schaffitzel et al., 2020).

Declaration of Competing Interest

The authors declare no competing interests.

Data availability

The data is public available. The data sources are referenced.

Appendix. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2023.107815.

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