Socio-economic impacts of current products and practices, incl. non-accounted externalities

Deliverable 7.1 – D29 – WP7

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OPTIMISING BIO-BASED FERTILISERS IN AGRICULTURE – PROVIDING A KNOWLEDGE BASIS FOR NEW POLICIES

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Dem	Demonstrator	
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LEX4BIO aims to reduce the dependence upon mineral/fossil fertilisers, benefiting the environment and the EU's economy. The project will focus on collecting and processing regional nutrient stock, flow, surplus and deficiency data, and reviewing and assessing the required technological solutions. Furthermore, socioeconomic benefits and limitations to increase substitution of mineral fertiliser for BBFs will be analysed. A key result of LEX4BIO will be a universal, science-based toolkit for optimising the use of BBFs in agriculture and to assess their environmental impact in terms of non-renewable energy use, greenhouse gas emissions and other LCA impact categories. LEX4BIO provides for the first-time connection between production technologies of BBFs and regional requirements for the safe use of BBFs.

The project runs from June 2019 to May 2024. It involves 20 partners and is coordinated by Luke (LUONNONVARAKESKUS - Natural Resources Institute Finland).

More information on the project can be found at: http://www.lex4bio.eu

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1. INTRODUCTION

1.1. Context and objectives

WP7 aims at higher use efficiencies of bio-based fertilisers (BBFs) and socioeconomic improvements for the rural population such as jobs, income and more liveable rural regions. Current fertilising practices and the use of BBFs can have different contributions to economic indicators (e.g. GDP, employment) and externalities (e.g. nitrate losses, ammonia emissions, resource depletion, biodiversity, soil health). In order to achieve the aim of WP7 and to develop sound, evidence-based policy recommendations it is paramount to understand how current fertilisation practices determine these economic contributions and the value of the negative externalities in different settings and regions. Providing this information is the objective of the present deliverable D7.1.

The following subchapters 1.2 and 1.3. give an overview of current production and use of mineral and organic fertilisers in the EU. Chapter 2 provides an explanation of some basic principles of valuing and monetising, which will be applied for a holistic assessment of socio-economic impacts of current fertilising practices across the EU in Chapter 3. Chapter 4 will analyse specific impacts in their regional context in three hotspot regions covered by LEX4BIO, i.e. legacy impacts of over-fertilisation in regard to eutrophication of the Baltic Sea (Chapter 4.1), health impacts associated to nitrogen emissions from mineral and organic fertiliser use in Flanders (Chapter 4.2) and inferences of fertilisation impacts that can be drawn from the effectiveness of agri-environmental programmes in two Austrian regions (Chapter 4.3). Perspectives from different stakeholders in the case study regions are collected in Chapter 4.4. The main insights gained through both EU-level and case study assessments are summarised in Chapter 5.

1.2. Current production and use of mineral fertiliser in the European Union

According to Fertilizer Europe (2020) in 2018 18.1 million tonnes of nutrients were produced in more than 120 fertiliser production sites across the EU-28, of which 74% were N, 11% P_2O_5 and 15% K_2O. Whereas import dependency for K_2O (71%) and P_2O_5 (66%) was high, only 28% of consumed N stemmed from imports. Main trade partners were Russia (imports worth EUR 1.52 billion), Egypt (EUR 0.49 billion), Belarus (EUR 0.46 billion) and Morocco (EUR 0.41 billion).

Total EU-28 fertiliser consumption in 2019 was 20 million tonnes of nutrients of which 17.3 million tonnes were used in agriculture, namely 11.2 million t N, 2.7 million t P_2O_5 and 3.1 million t K_2O. 75% of the agricultural area is fertilised, with wheat (26%) and coarse grains (25%) accounting for more than half of agricultural fertiliser consumption (Fertilizer Europe 2021). Regarding N fertiliser, nitrates were with 46% the most popular form of fertiliser consumption, followed by urea (21%), UAN and compound fertilisers (13% each). This is a clear difference to the global scale, where urea (48%) and compound fertilisers (20%) were the most widely applied forms of N fertiliser (Fertiliser Europe 2020).

After a steep reduction in the early 1990s, fertiliser use continued to decrease more slowly up to the economic downturn in 2008 followed by a slight recovery over the last decade. For the growing season 2029/2030 expected changes are -6%, -2.1% and +0.9% for N, P_2O_5 and K_2O, respectively. Over the same period a decrease in agricultural area of 1% is projected (Fertiliser Europe 2021).

Figure 1 shows current fertilisation rates and changes in fertiliser application for the different EU countries. N fertilisation rates are particularly high in Western Europe and tend to decrease towards the East and South, whereas the opposite is true for P fertiliser. For all three nutrients highest growth rates are reported in Bulgaria and Romania. It should be noted that the period for comparison (growing seasons 2005/06 to 2008/09) partly fell into a period of economic crisis accompanied by particularly low fertiliser use. On absolute terms, four countries i.e., France, Germany, Poland, and Spain account for approximately half of European fertiliser use.

1.3. Current use of organic fertiliser in the European Union

In 2014-2017 on average 9.5 million tonnes N and 1.6 million tonnes P in the form of manure were applied to agricultural land per year. This constitutes a decrease of 4% for N and 6% for P compared to the period 2004-2007. The majority of manure (61% of N and 53% of P) stems from cattle (Eurostat 2021a). Manure application rates, as shown in Figure 2, reflect patterns of livestock density (Eurostat 2020a) and are highest in the Netherlands, Malta, and Cyprus and lowest in Latvia, Bulgaria, and Lithuania. Regarding Cyprus, sheep and goat manure is the dominant form of manure though. Most countries report no imports of manure or withdrawals to other sectors than agriculture, except for France where more than 20 000 t of both N and P were imported in 2016 and 2017 (Eurostat 2021a). However, the datasets on manure imports and withdrawals are incomplete and mass balances do not add up so that it is not possible to draw a valid conclusion.

Organic fertilisers other than manure (e.g. sewage sludge and compost) only play a subordinate role (Eurostat 2021a). The average annual application in the period 2014-2017 was 0.4 million tonnes N. For P data reporting is not complete.

No data is available on the application of K with organic fertiliser.

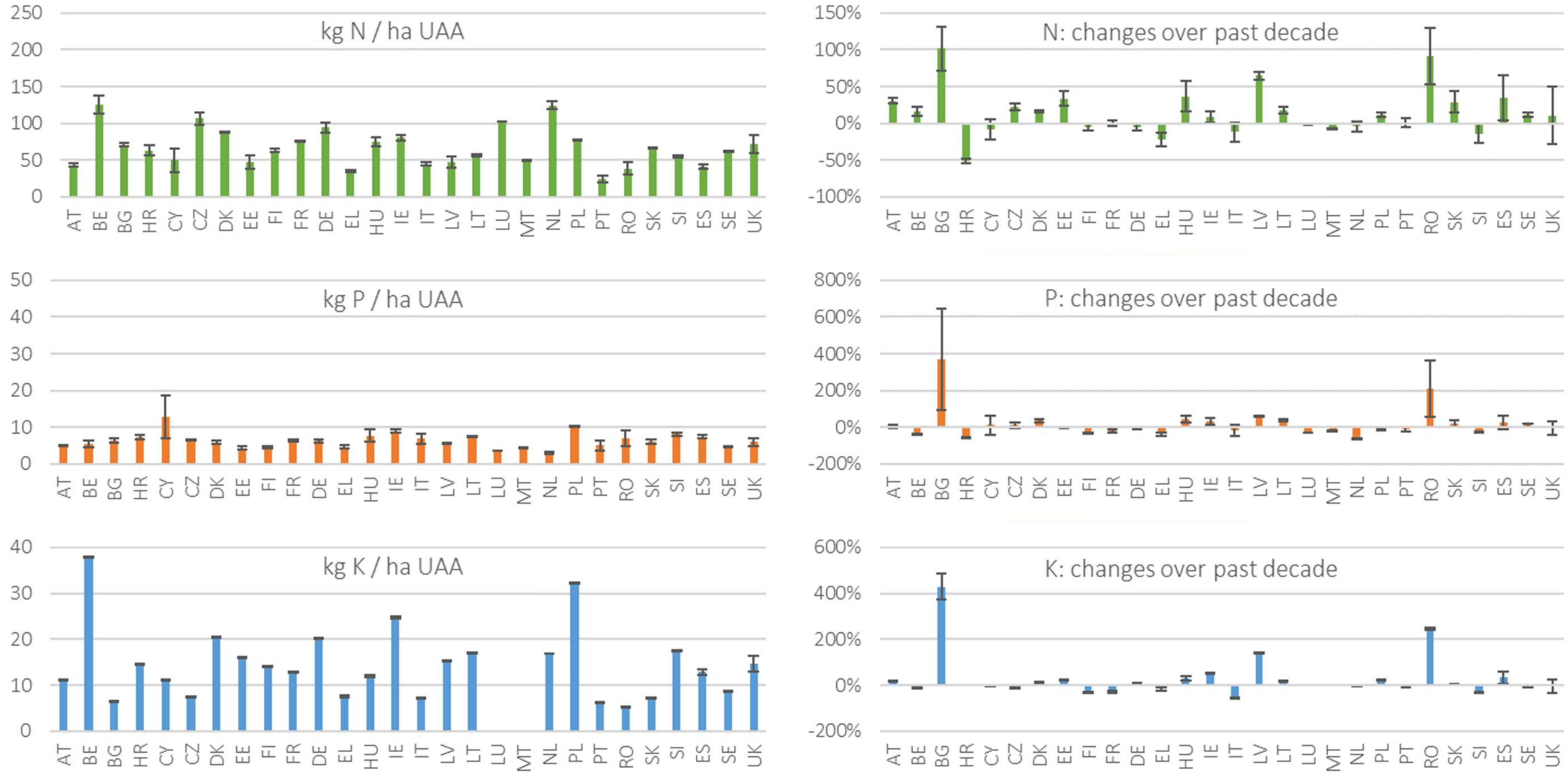

Figure 1: Current fertiliser use in the European Union (average over the growing seasons 2016/17 to 2018/19) and changes in fertiliser application over the last decade (comparison of growing seasons 2016/17 to 2018/19 and 2006/07 to 2008/09). For K data for LU are included in BE, no data available for MT. UAA: utilised agricultural area. Error bars refer to differences in the Eurostat datasets. Data on fertiliser refer to both sales and use of mineral (manufactured) fertiliser (Eurostat 2021 a-ap).

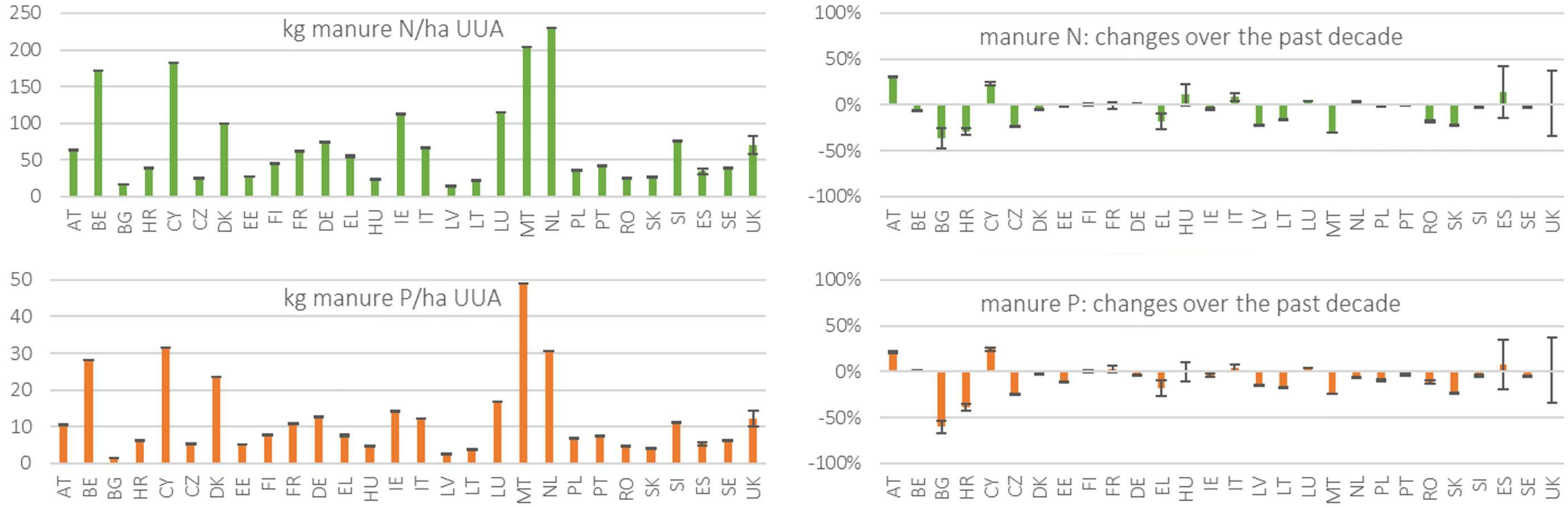

Figure 2: Current manure application in the European Union (average of 2014-2017) and changes in manure application over the last decade (comparison of the periods 2004-2007 and 2014-2017). UAA: utilised agricultural area. Error bars refer to differences in the Eurostat datasets on UAA (Eurostat 2021 a-ap).

2. METHODOLOGY

In the following sections, economic, environmental and social impacts of current fertiliser production and use will be analysed. The focus is on impacts within Europe, however, where relevant environmental and social impacts occur along the supply chain outside Europe, as is e.g. the case for phosphate mining (Chapter 3.2 and 3.5), these are also considered. Where no data specific for Europe could be obtained, e.g. for contribution of fertilisation to food security (Chapter 3.6.1), impacts are analysed on a global scale.

Environmental impacts can lead to other environmental, social and economic impacts. For better readability, these impacts are described together with the main impacts, e.g. economic losses in coastal tourism due to marine eutrophication will be discussed under environmental impacts of N and P runoff (Chapter 3.3.4)

2.1. Quantifying impacts

Where possible, impacts are quantified not only as emissions but also as the effects they have on human physical and mental well-being, commercial activities, species abundance and diversity, etc. An exception are impacts on climate change which are only described as greenhouse gas emissions and in monetary terms, because impacts of climate change are so extensive that their holistic description is beyond the scope of this report and because they are already accounted for in existing monetisation methods. Data for quantifying different impacts stem from scientific articles, statistical databases, market reports and standards in common environmental assessment methods and are described in the respective sections. Where data availability and/or scientific understanding do not yet allow for quantification of an impact, a qualitative description is given.

2.2. Monetising impacts

There are multiple approaches to monetise environmental, economic and social impacts of an activity. For instance, economic impacts can be assessed by comparing the value (expressed e.g. as monetary profit or employment) generated by using (and polluting) the environment with the benefits foregone, if it deteriorates.

Stated and revealed preference methods, on the other hand, have established themselves for measuring human well-being. This poses the challenge of quantifying impacts that cannot be observed from market prices. Revealed preference methods try to solve this problem by using observations from related markets, such as distances people travel to visit recreational sites and the associated costs, or the real estate values in a certain area. However, as prices on related markets may be influenced by a variety of factors, sufficient uncorrelated control variables and independent observations have to be included (Sagebiel et al. 2016). Stated preference methods measure people's willingness to pay (WTP) for certain ecosystem services based on surveys in which respondents are confronted with hypothetical decision situations. However, it is only with great caution that conclusions can be drawn from stated preference methods. Results are highly dependent on the survey design and thus on factors such as the proposed timeframe and mode of payment, whether questions are open-ended or closed-ended, method of surveying (e.g. online panels or personal interviews) and the treatment of non-responses or zero-responses. For instance, non-responses can be either neglected in the evaluation or treated as zero willingness to pay. "Protest responses" where people have constraints preventing them from stating their true willingness to pay constitute further sources of uncertainty

(Sagebiel et al. 2016). Such constraints include, among others, opposition to the proposed payment method, e.g. respondents may prefer polluter-pay approaches over common taxes (BalticSTERN Secretariat 2013). Nevertheless, stated preference is to date the only method able to include non-use (or existence) values of an ecosystem into the assessment (HELCOM 2018a).

Another commonly applied approach is to analyse (real or hypothetical) abatement costs to mitigate the effects of activities causing environmental degradation. Abatement costs to prevent negative effects from arising are also often calculated, however, these are mostly used for comparison in cost-benefit analyses rather than for valorisation of the impacts themselves.

As a holistic assessment should include all different aspects of an impact, estimates derived with different approaches are included in the present assessment. However, although all results are expressed in monetary terms it is not possible to make a direct comparison or overall sum. For one, the calculation methods and approaches for different effects differ (HELCOM 2018a). Secondly, results of different valuation methods may overlap. For instance, the value people assign to the preservation of an ecosystem may partly be influenced by the economic profit they gain from using its services (BalticSTERN Secretariat 2013). Furthermore, it is not possible to derive a monetary value for all impacts. The focus of the present assessment was on monetising those impacts that are likely to cause the biggest damage as uncertainty ranges in monetised impacts are high so that the exclusion of minor impacts is not likely to change the conclusions of the study (Sutton et al. 2013).

3. SOCIO-ECONOMIC IMPACTS OF CURRENT FERTILISING PRACTICES ACROSS THE EU

3.1. Economic impacts of mineral fertiliser production

Fertiliser plants can be found in all EU member states except for Cyprus, Luxembourg and Malta. Nevertheless, domestic production has declined rapidly in the 1990s and early 2000s, as seen in Figure 3 for the examples of France and Italy, whereas for the total manufacturing sector hardly any changes can be observed (Eurostat 2021ax). However, over the last decade, the fertiliser industry has remained relatively stable around an annual production volume of EUR 15 billion and a contribution of 0.13% (EUR 9.8 billion including supply chains) to the turnover of the manufacturing sector (Eurostat 2021ay, Eurostat 2021az, Fertilizer Europe 2020).

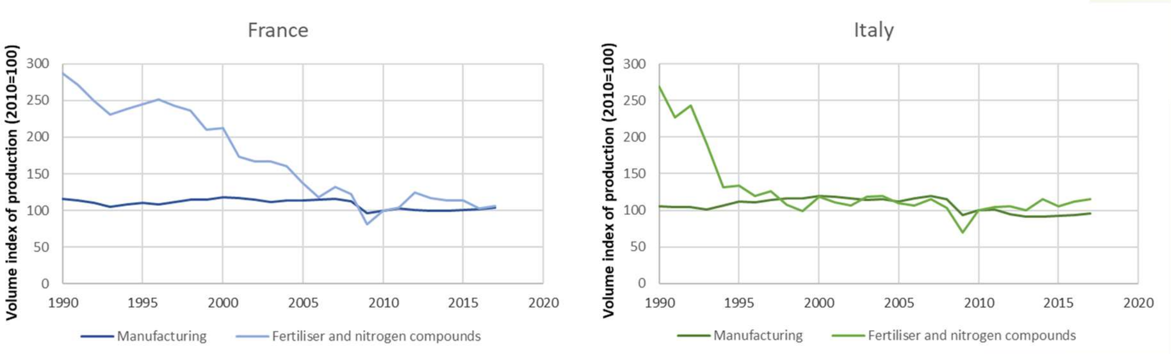

Figure 3: Volume index of production for manufacture and manufacture of mineral fertiliser and nitrogen compounds over time in France and Italy (Eurostat 2021ax).

Except for organic fertilisers, which play a minor role in the fertiliser industry, all fertiliser types show a negative trade balance (see Figure 4). Overall, fertilisers worth EUR 4.3 billion and EUR 2.9 billion are imported and exported each year, respectively, corresponding to 0.2% and 0.1% of the total import and export volumes of the EU. Although both import and export volumes have increased over the last decade, the share of fertilisers in total import value remained constant at 0.2%, whereas their share in total export value was halved (from 0.2% to 0.1%) between 2007-2009 and 2017-2019. In addition to trade with external countries, there is extensive trade among the EU member states with a volume of EUR 7-8 billion (0.2% of the total internal trade). Overall, six countries (Belgium, Croatia, Germany, Lithuania, Netherlands and Slovakia) exhibit a positive trade balance, as shown in Figure 5. Belgium and the Netherlands contribute most to the total export volume, whereas France has the highest import of fertiliser (see Figure 5). However, compared to the total export volume, fertiliser play with 2.7% a particular important role in Lithuania.

Figure 4: Import and export volumes of different fertiliser types to/from the European Union. Average of the respective seasons and of different Eurostat datasets (Eurostat 2021ay, bb-be).

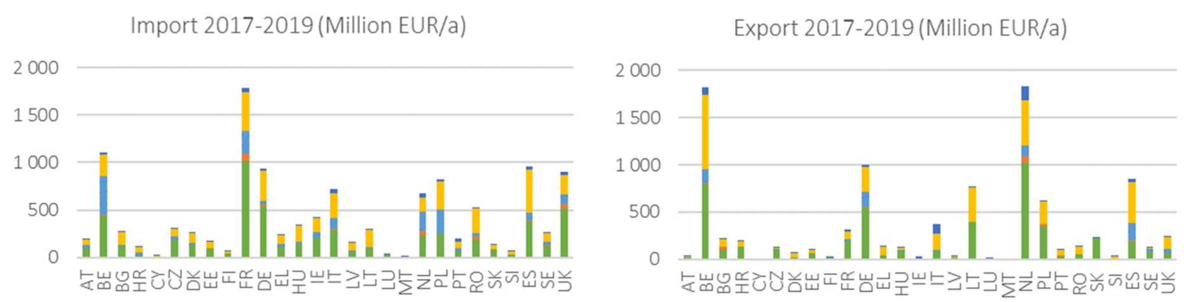

Figure 5: Import and export volumes of different fertiliser types by EU country. Average of the respective seasons and of different Eurostat datasets (Eurostat 2021ay, bb-be).

Regarding the labour market, the fertiliser manufacturing sector engages 75 000 employees (including supply chains), which is 0.03% of the European workforce (Eurostat 2021ay, Fertiliser Europe 2020). There is no data on wages available specifically for the fertiliser manufacturing sector. However, wages and salaries in the manufacture of chemical and chemical compounds, of which the fertiliser industry is a part, are with EUR 3195 per month per full-time employee (excluding apprentices) above the average wages in industry, construction and services (EUR 2381) and have been growing more rapidly (18% compared to 7%) between 2012 and 2016 (the last year, for which data is available; Eurostat 2021bf). This is also true for apprentices' wages and salaries, although differences are smaller (EUR 1517 per full-time employee per month compared to EUR 1275 in industry, construction and services in 2016) and fluctuations of wages higher than in other economic sectors.

3.2. Economic impacts of fertiliser use

Agriculture contributes with EUR 181 billion (1.3%) to the GVA (gross value added) of the EU (average 2016-2018). Although the agricultural GVA increased by 5% on absolute terms and 8% on a per ha basis over the past decade, its share of the total European GVA decreased by 0.1% (Eurostat 2021aq,au). With more than 4% agriculture contributes a comparatively high part to total GVA in Bulgaria and Romania, whereas it accounts for less than 0.5% of the total GVA in Sweden and Luxembourg. However, Bulgaria and Romania are also the countries that experienced the highest decrease in agriculture's share of total GVA over the past decade (see Figure 6).

The annual labour input in the agricultural sector was around 9 million AWU (annual work units, equivalent to 1800 h/year) in the period 2016-2018; a reduction of 22% compared to 2006-2008. Although the share of salaried work has been increasing over the last decade, 74% of the labour input in the agricultural sector is still non-salaried (Eurostat 2021at). Labour input per generated output tends to be higher in Eastern European countries with Romania having the highest labour input (104 AWU/million EUR agricultural output) and Denmark and the Netherlands the lowest (6 AWU/million EUR agricultural output). However, on a per area basis, labour input is with 0.44 AWU/ha UAA highest in Malta (see Figure 6). Compensation of employees in the agricultural sector was EUR 10 per hour in the period 2016-2018 and is thus 58% below the average across all economic sectors. Differences between the member states are high though, with compensation for agricultural employees amounting to more than 70% of the average in Slovakia, Poland, the Czech Republic, Ireland, Bulgaria, and Hungary, but only to less than 40% of the average in Sweden, Spain, Luxembourg, and Malta (Eurostat 2021aq,at,av,aw).

Annual expenses for fertilisers in the EU amounted to EUR 17 billion on average for the period 2016-2018. France, Germany, Spain and Poland, who together consume half of mineral fertilisers (see above), are also responsible for more than half of fertiliser expenses (Eurostat 2021aq). Prices are highly variably between different countries and fertiliser types. While mineral fertiliser prices are also highly variable on a temporal scale with fluctuations of 37% over the last decade, prices for organic fertilisers and soil improvers have remained comparatively constant since 2010 (Eurostat 2021ar,as). Expenses per agricultural areas tend to be higher in countries with high N-fertiliser application rates, as shown in Figure 7, although Cyprus has particularly high, and the Czech Republic particularly low fertiliser expenses compared to their respective application rates. However, fertilisers make up less than 10% of total intermediate agricultural consumption in most countries (7% on EU average). Furthermore, fertiliser expenses have probably, but not significantly, increased to a lower degree over the last decade (7%) than both crop output (8%) and most other types of intermediate consumption (Figure 7). Over the same period crop production volume has increased 8% whereas fertiliser volumes decreased by 1% (Eurostat 2021bg). Thus, the price ratio between crop and fertiliser increased by 8%.

Overall, it can be said that the agricultural sector is a small contributor to the total EU economy and its importance is declining over time. Fertiliser expenses only make up a small part of intermediate consumption and are growing more moderately than monetary crop output. In the INCA project (Vysna et al., 2021) the contribution of human input to crop yield is estimated with 79%. Considering the share of fertilisers in intermediate consumption and labour compensation, fertilisation would be responsible for 4% or EUR 7 billion of agricultural GVA. Yet, yield benefits of fertilisation are highly dependent on crop type, soil and climatic conditions. For instance, unit benefits of fertilisation are high for vegetables, whereas root crops require less fertilisation. Furthermore, economic benefits of organic fertilisers have hardly been assessed (Brink et al 2011). The amount of output and consequently maintenance of employment in the sector that can be directly related to the use of fertilisers can only be quantified by bioeconomic modelling.

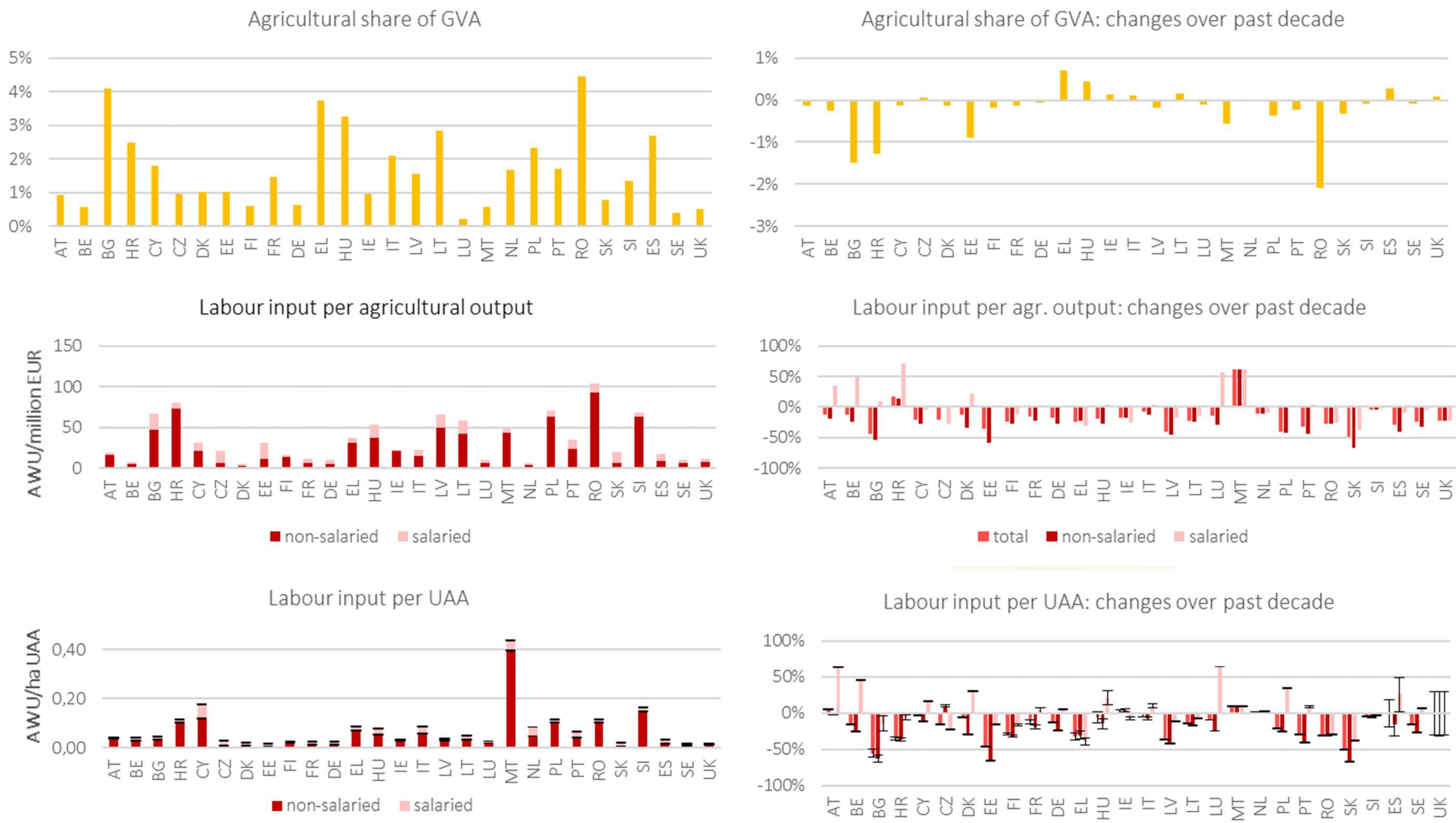

Figure 6: GVA and current labour input in the agricultural sector in the European Union (average of 2016-2018) and changes over the past decade (comparison of the periods 2006-2008 and 2016-2018). GVA: gross value added. AWU: annual work unit. UAA: utilised agricultural area. Error bars refer to differences in the Eurostat datasets on UAA (Eurostat 2021 a-ap,aq,at,au).

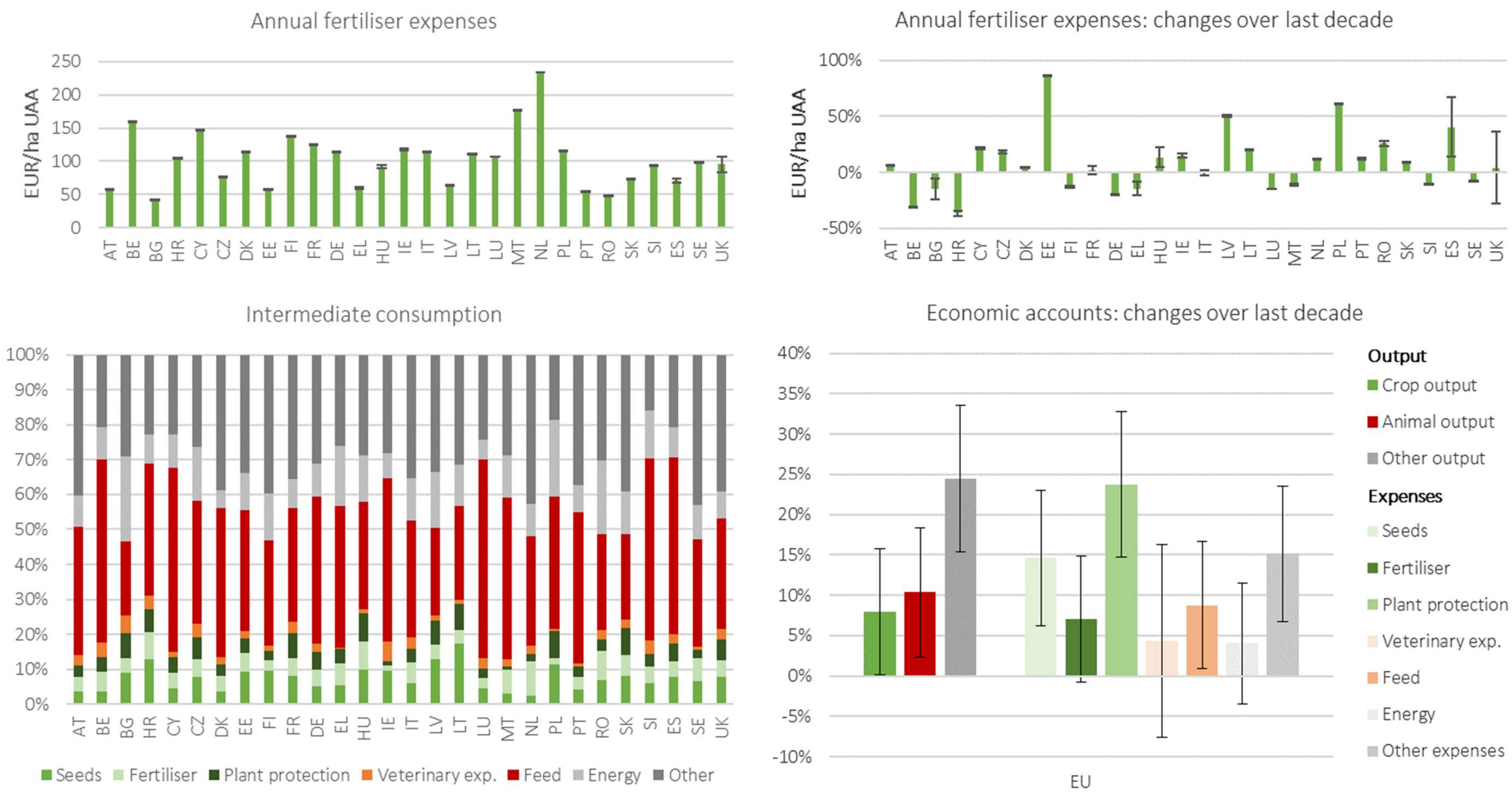

Figure 7: Current fertiliser expenses (at basic prices) in the European Union, share of total intermediate consumption of the agricultural sector (average of 2016-2018) and changes over the last decade (comparison of the periods 2006-2008 and 2016-2018; inflation-adjusted). UAA: utilised agricultural area. Error bars refer to differences in the Eurostat datasets on UAA (Eurostat 2021 a-aq).

3.3. Environmental impacts of fertiliser use

3.3.1. Impacts of greenhouse gas emissions

Approximately 1% of reactive N applied with mineral fertiliser and manure is emitted to the environment as N_2O. In addition, N_2O and CH_4 emissions occur during manure management, mainly storage (EEA 2021). N_2O and CH_4 are potent greenhouse gases with a radiative forcing of 273 ± 130 and 27 ± 11 times that of CO_2 (Forster et al. 2021). According to the latest National Inventory Report of the EU (EEA 2021) direct N_2O emissions from mineral fertiliser use in 2019 amount to 50.46 Mt CO_2 equivalents (CO_2eq). More than half of emissions stem from France (20%), Germany (13%), Spain and Poland (9% each). However, France and Germany are also among the countries that have achieved the highest absolute decrease in emissions since 1990. On the other hand, German N_2O emissions from organic fertiliser use have increased considerably, making it the highest contributor to EU emissions in that category (23%), followed by France (11%) and Italy (10%). Overall, direct emissions from organic fertiliser were 25.23 Mt CO_2eq. CH_4 and N_2O emissions during manure management equal 40.56 Mt CO_2eq and 13.82 Mt CO_2eq, respectively. CH_4 emissions were highest in Spain (17.1%), Germany (14.4%) and Italy (10.2%), whereas the United Kingdom (17%), Germany (14%), France and Poland (10% each) account for the highest share of N_2O emissions. However, emissions from manure management have been significantly reduced since 1990 (by 18% for CH_4 and by 37% for N_2O). Together, direct greenhouse gas emissions from fertilising make up 30% of agricultural and 3% of total greenhouse gas emissions of the EU (EEA 2021). CO_2 emitted from machinery and vehicles used for fertiliser application have not been included in these figures.

Global warming is probably the environmental impact for which monetisation is currently most advanced. 12 EU member states collect carbon taxes ranging from less than EUR 1 per t CO_2 in Poland to EUR 109 per t CO_2 in Sweden (as of 01.04.2020, World Bank Group 2020). Emission allowances are also traded via the EU emission trading system (ETS), where prices have increased steadily since the latest reform in 2018 and currently hold at EUR 54 per t CO_2 (average of the period 05.06.2021 – 20.08.2021, Finanzen.net GmbH n.d., EMBER 2021). However, both carbon taxes and the ETS only cover a specific set of sectors and fertiliser application is not included in any of these systems. Moreover, market prices of carbon do not necessarily correspond to the “true” social and environmental costs. For instance, the International Monetary Fund estimates that average global carbon price is EUR 2 per t CO_2, whereas EUR 67 per t CO_2 would be needed to limit global warming to 2°C (Parry 2019). Similarly, according to the Carbon Pricing Leadership Coalition (CPLC 2017) prices in the ETS should have already reached EUR 35-71 per t CO_2 in 2020 and reach EUR 44-89 per t CO_2 by 2030 in order to remain on track for the goals under the Paris Agreement. Other studies have estimated social costs of carbon via willingness to pay (WTP) surveys (van Grinsven et al. 2013) or macroeconomic damage functions (Ricke et al. 2018, IWGS 2021). While the estimates by van Grinsven et al. (2013) and the IWGS (2021) are with EUR 8000-34 000 per t N_2O (equivalent to EUR 29-125 per t CO_2) and EUR 5085-23 674 per t N_2O (equivalent to EUR 19-87 per t CO_2) in a similar range, Ricke et al. (2018) find negative carbon costs in all of Europe except for Greece, Italy, Portugal and Spain, at least in some of the analysed scenarios. Nevertheless, global median costs are with EUR 362 per t CO_2 (EUR 177-805 per t CO_2 at 66% confidence interval) higher than in the other two studies. Considering only global estimates and ETS prices deemed adequate to reach international climate goals, social costs of greenhouse gas emissions from fertilising in the EU in 2019 range between EUR 2.6 billion and EUR 104.7 billion.

In addition to direct global warming potential N_2O also contributes to the decrease in stratospheric ozone (O_3). This on the one hand increases radioactive forcing and thus contributes to global warming and on the other hand increases risk for skin cancer and potentially cataract. However, whether effects of N_2O on stratospheric O_3 depletion should be considered in environmental assessments is disputed, as the governing chemical processes are to date not well understood (Butterbach-Bahl et al. 2011, Verones et al. 2020). Veronese et al. (2020) estimate 0.002-0.027 disability adjusted life years (DALY) per t N_2O emitted, i.e. life years lost due to premature death or impaired by sickness. For EU 2019 emissions this would equal 444-6259 DALY. The value of a lost life year in the EU is quantified between EUR 57 700 – 138 700 (Holland 2012). In view of the large range, differences between years lived with a disability and life years lost can be neglected (van Grinsven et al. 2010) and total health costs of reactive N emissions amount to EUR <0.1 – 0.9 billion for 2019 emissions. WTP-based cost estimates in van Grinsven et al (2013) are EUR 1-3 per kg N_2O-N emitted, amounting to an equal range of EUR 0.1-0.9 billion.

3.3.2. Impacts of NH_3 and NO_2 emissions

In 2018 agricultural NH_3 and NO_2 emissions of fertiliser application and manure management were 3.38 Mt and 0.51 Mt, respectively. In addition, 0.31 Mt NO_2 were generated during storage, handling and transport of agricultural products as well as off-road machinery use, part of which may also be associated with fertiliser use. Fertilisation practices thus account for 88% of European NH_3 emissions and 7-11% of NO_2 emissions. (Eurostat 2021bh). Absolute emissions were highest in Germany and France, whereas related to UAA NO_2 emissions were highest in the Netherlands (15.5-18.2 kg/ha) and lowest in Portugal (0.6-1.9 kg/ha) and NH_3 emissions were highest in Malta (103.3 kg/ha) and lowest in Latvia (6.4 kg/ha) as shown in Figure 8.

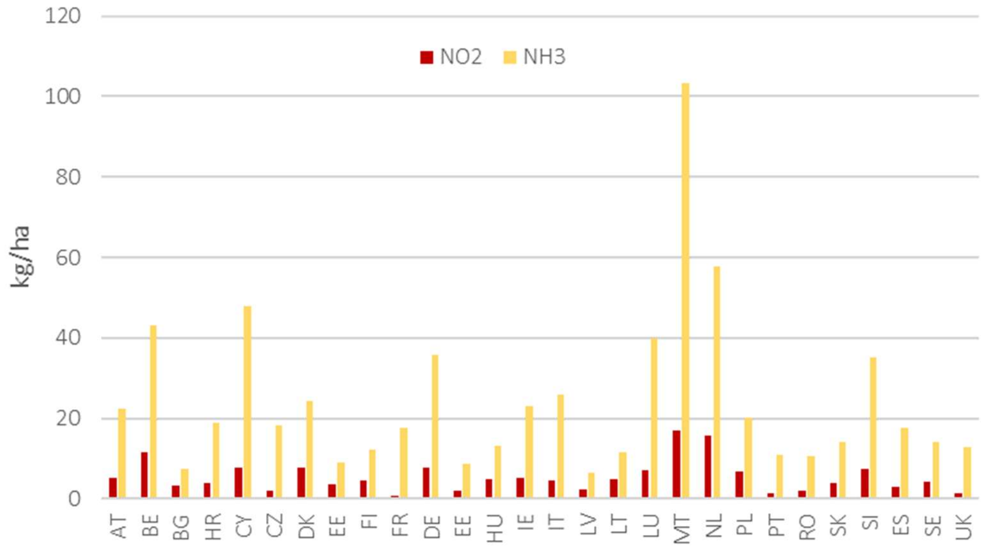

Figure 8: NO_2 and NH_3 emissions from fertiliser use in the EU in 2018. Datasets on NO2 emissions for France and Czech Republic are incomplete and emissions may be underestimated. UAA: utilised agricultural area (Eurostat 2021a,am-ap,bh).

While being toxic on their own, NH_3 and NO_2 are also precursors of particulate matter (PM). In addition, NO_2 amplifies the effects of PM and contributes to the formation of ground level ozone (O_3). PM, NO_x and O_3 are the most important air pollutants in the EU, causing respiratory and cardiovascular diseases and being responsible for 863, 120 and 46 years of life lost per 100 000 inhabitants in 2018, respectively (Moldanová et al. 2011, EEA 2020). Environmental impact assessments typically take both mortality and morbidity into account and measure negative effects on human health as disability-adjusted life years (DALY). Commonly applied or recently recommended methods in the EU quantify

$PM_{2.5}$ effects between 0.004-0.105 yr/t NO_x and 0.014-0.271 yr/t NH_3 (UNEP 2016, Fazio et al. 2018, Verones et al. 2020). Mortality and morbidity estimates for O_3 caused by NO_x emissions range between 0.00004-0.00031 DALY/t (Fazio et al. 2018, Verones et al. 2020). Applying costs per DALY as in Chapter 3.3.1, total health costs of reactive N emissions amount to EUR 2.8-139.0 billion. The WTP-based estimate reported by van Grinsven et al. (2013) of EUR 10-30 per kg NO_x-N emitted and EUR 2-20 per kg NH_3-N emitted, yields damage costs of EUR 7.1-63.1 billion for 2018. The German Environmental Agency (Matthey and Bünger 2020) suggests EUR 15.2 per kg NO_x emitted and EUR 22.8 per kg NH_3 emitted yielding a total of EUR 84.8-89.5 billion. However, German values refer to the national situation and may not be applicable for the whole of Europe.

In addition to their impacts on human health NO_2 and NH_3 also influence climate and plant growth. Both substances form aerosols, for which van Grinsven et al. (2013) estimate cost of EUR -9-2 per kg NO_x-N emitted and EUR -3-0 per kg NH_3-N emitted. However, while at the time van Grinsven et al. (2013) conducted their analysis it was yet unclear whether aerosols have an overall warming or cooling effect, the latest IPCC report (Foster et al. 2021) confirms a negative effective radiative forcing of -2.0 to -0.6 W/m². Therefore, aerosol formation is likely to have reduced global warming costs of current fertilising practices by up to EUR 10.60 billion in 2018. Moreover, through scattering of light, aerosols increase diffuse radiation which may increase ecosystem production and thus carbon storage (Butterbach-Bahl et al. 2011). On the other hand, as a precursor of O_3 NO_2 also amplifies global warming. However, due to the complex interaction of different factors governing O_3 formation as well as several positive and negative feedbacks the relation between NO_2 emissions and increased radiative forcing is difficult to quantify (Butterbach-Bahl et al. 2011). Furthermore, O_3 damages plant tissues, causing losses in crop productivity and inhibiting ecosystem C storage (Sutton et al. 2013, Butterbach-Bahl et al. 2011). According to van Grinsven et al. (2013) crop damage amounts to EUR 1-2 per kg NO_x emitted, i.e. EUR 0.25-0.50 billion for 2018 emissions. This is well in line with marginal damage costs for each country in the CAFE report (Holland et al. 2005) ranging from EUR -95 per t NO_x in the Netherlands to EUR 500 per t NO_x in France and the estimation by Sutton et al. (2013) that 60% of O_3 concentration increase can be attributed to NO_x emissions and O_3 damage to plants causes 5% losses in agricultural crop productivity. The latter two assumptions yield EUR 0.11-0.20 billion and EUR 0.48-0.75 billion for EU emissions in 2018, respectively (Eurostat 2021aq,bh). Regarding carbon sequestration, it is estimated that O_3 induced damage to vegetation has reduced carbon sequestration by up to 5-14 Mt C per year between 2000-2005 (Butterbach-Bahl et al. 2011). Considering the cost range of carbon discussed in Chapter 3.3.1, this equals EUR 0.18-11.27 billion.

Furthermore, increased emission of reactive N also leads to its increased deposition as gas, in dry form or with precipitation. While N deposition can be considered as an additional form of fertilisation in crop systems, it is not targeted to growth periods or crop needs and may thus increase nutrient leaching and runoff. In natural ecosystems increased N deposition may change species composition, increase susceptibility to abiotic (e.g. drought and frost) and biotic (e.g. attacks by insects and fungal pathogens) stresses, affect plant palatability and increase allergenic pollen production (Moldanová et al. 2011, Dise et al. 2011, Sutton et al. 2013). In Europe forests on nutrient poor soils, semi-natural grasslands, ombrotrophic bogs and nutrient-poor fens, heathlands, sand dunes, and species such as sundew, bryophytes, lichens, funghi and forb are especially vulnerable to N deposition. Changes in the vegetation composition in turn change the composition of the fauna associated with that vegetation (Dise et al. 2011). However, soil fauna diversity may benefit from nutrient enrichment, as it seems to be more sensitive to the quantity than the quality of available organic matter. An increase in earthworm biomass could in turn have effects on soil aggregation, water infiltration and organic matter dynamics (Velthof et al. 2011). Also transmitters of several parasitic and infectious human and livestock diseases seem to be well adapted to nutrient rich environments (Sutton et al. 2013). Another problem associated with N deposition is soil acidification, i.e. a decrease in soil pH causing deficiencies of essential plant nutrients such as P, calcium, magnesium and molybdenum, release of toxic compounds (especially aluminium), as well as accumulation of ammonium and litter (Velthof et al. 2011, Dise et al. 2011). In 2018 average N deposition in Europe amounted to 1.2 g/m² (EMEP 2020), 54-57% of which can be attributed to fertiliser emissions. Critical loads for eutrophication have been exceeded in 64.8% of ecosystem area. Critical acidification loads, although not only caused by reactive N, but also sulphur deposition, have been exceeded in 5.2% of ecosystem area (EMEP 2020). Of the commonly applied impact assessment methods in Europe only Goedkoop et al. (2009) provide endpoint characterisation factors. Depending on the time horizon considered, each t of NO_x emissions causes disappearance of $0.69*10^{-6}$-$1.01*10^{-5}$ species and each t of NH_3 between $1.52*10^{-6}$-$1.42*10^{-5}$ species. 2018 fertilising emissions have thus caused the disappearance of 5-56 species from EU ecosystems. For eutrophication impacts to our knowledge no similar estimations have been made to date. However, in a global review study, Soons et al. (2017) found a 16% reduction in species richness of herbaceous vegetation for N deposition rates of 0.4-60 g/m². According to Sutton et al. (2013) 5-10% of global biodiversity loss can be attributed to enhanced N deposition and the OECD (Sud 2020) estimates that NH_3 and NO_x emissions have reduced forest biodiversity by more than 10% over two thirds of Europe. Cost estimates for terrestrial eutrophication and biodiversity impacts of NH_3 and NO_x emissions based on WTP have been made by van Grinsven et al. (2013) and amount to EUR 2-10/kg reactive N emitted. This yields EUR 5.9-30.3 billion for 2018 emissions in the EU. Matthey and Bünger (2020) quantify impacts on biodiversity with EUR 2.8 per kg NO_x and EUR 11 per kg NH_3 for Germany, i.e. EUR 373.2-374.1 billion if applied to the whole EU. However, acidification not only affects ecosystems but also damages buildings and other structures via enhanced corrosion (Moldanová et al. 2011). Holland (2012) estimates the combined benefits on materials and crops when reducing air pollution to the maximal technically feasible level between 2012 and 2030 with EUR 0.80 billion. Given the cost estimates for crop damage of EUR 0.25-0.75 billion it can be concluded that material damage is a rather small contributor to the overall emission costs.

As a secondary effect, N deposition and soil acidification increase N_2O emissions from soil. These indirect N_2O emissions were 9.3 Mt CO_2eq in 2019 (EEA 2021), of which 5.0-5.3 Mt CO_2eq can be attributed to fertilising. Applying the same premises as in Chapter 3.3.1 the equivalent costs amount to EUR 0.2-4.3 billion. On the other hand, N deposition enhances carbon sequestration. For forests and heathland for instance, C uptake is estimated with 5-75 g/g N deposited (Butterbach-Bahl et al. 2011). Given an area of forests, shrubland and spontaneous (grass) vegetation of 2 094 285 km² (Eurostat 2021bi) benefits for carbon sequestration in natural ecosystems are EUR 0.4-151.7 billion. Furthermore, effects of increased CH_4 emissions and increased albedo of vegetation due to increased N deposition (Butterbach-Bahl et al. 2011) have not been quantified.

Similarly, N deposition generally increases N leaching at rates above 1 g N/m², i.e. at rates below the EU average N deposition (Velthof et al. 2011). However, N leaching rates from natural land are rarely measured and a distinction of leaching due to N deposition is complex so that this impact could not be quantified. Thus, increased N_2O emissions from waterbodies following deposition induced N leaching have not been considered either.

3.3.3. Impacts of NO_3 leaching to groundwater

Average nitrate (NO_3) concentration in European groundwater amounted to 21.98 mg/l in 2018. The average annual increase over the last decade was 0.05 mg/l (Eurostat 2021bj). Besides average concentrations local distribution is important. Both WHO guidelines and the EU drinking water directive include limit values for drinking water of 50 mg NO_3/l (Directive (EU) 2020/2184, WHO 2017). Exposure to higher levels increases the risk for infant methaemoglobinaemia and damage to thyroid functions (WHO 2016). Recent evidence on other risks such as colorectal cancer and central nervous system birth defects already at lower exposure levels are yet insufficient to allow for firm conclusions (WHO 2016, Ward et al. 2018). Furthermore, in pyrite containing soils, which are e.g. common in Denmark, England, France, Germany, the Netherlands and Spain, NO_3 increases pyrite oxidation and consequently concentration of heavy metals, sulphate and other drinking water contaminants in groundwater (Velthof et al. 2011).

In 2012–2015, 13.2% of groundwater stations in the EU exceeded 50 mg NO_3/l and 5.7% were between 40 and 50 mg/l. Exceedance rates were highest in Malta (71%), Germany (28%) and Spain (21.5%) and lowest in Ireland (0%), Finland (0.5%) and Sweden (0.9%). However, member states apply different monitoring strategies and NO_3 concentrations are also dependent on geological and climatic conditions (European Commission 2018a,b). NO_3 pollution not only stems from leaching after application of mineral and organic fertiliser, but also from wastewater treatment, waste products and discharges from industrial processes and motor vehicles (WHO 2016). Nevertheless, agriculture is considered the main source. Of the 14 EU member states that reported the contribution of agriculture to N discharge into the aquatic environment (including surface waters) for the period 2012-2015 shares ranged between 62% in the Netherlands and 99% in Poland (European Commission 2018b).

WTP for avoiding negative health effects in drinking water according to van Grinsven et al. (2013) is EUR 0-4 per kg NO_3-N emitted. In an earlier study assuming a positive relationship between nitrate intake and colon cancer, van Grinsven et al. (2010) found damage costs for lost or disabled life years of EUR 0.7 per kg NO_3-N in the EU12 (except Greece). Neglecting NO_3 emissions to surface water and assuming 22% of N losses from agriculture occur as NO_3 (Velthof et al. 2013) a first rough cost estimation according to the range given in van Grinsven et al. (2013) yields up to EUR 7.50 billion. Agricultural N losses in this estimation are taken from the Gross Nutrient Balance (Eurostat 2021a) in 2014, the last year for which data is available. Yet another approach is to quantify NO_3 pollution costs with the expenses needed for drinking water treatment (e.g. Moore et al. 2011, Keeler et al. 2016, Lopes et al. 2019). However, these studies usually refer to specific municipalities or regions, while an EU-wide assessment of NO_3 removal costs is to date missing.

In addition, N leaching increases the N_2O release from waterbodies. A distinction between leaching and runoff from agricultural land is not made, however, together they accounted for emissions of 19.9 Mt CO_2eq (EEA 2021) or EUR 0.7-16.0 billion.

3.3.4. Impacts of N and P runoff

Although neglected in the previous section reactive N in agricultural runoff is likely to be significant in many regions (Butterbach-Bahl et al. 2011). Furthermore, part of the reactive N in soil and groundwater reaches rivers and streams via subsurface flow. P emissions, on the other hand, are almost exclusively associated with runoff and soil erosion. Excessive N and P emissions cause eutrophication of rivers, lakes and coastal areas with numerous consequences including shifts in species composition, reduction in species diversity, more frequent occurrence of bloom-forming, potential toxic algae and gelatinous zooplankton, odour and increased water turbidity. This in turn may affect the fishing and tourism industries as well as the real estate sector. Moreover, eutrophication can contribute to the spread of infectious diseases both directly by enhancing the replication rate of aquatic pathogens and indirectly by increasing abundance and distribution of their hosts and vectors. This is not only true for human pathogens, but also, e.g. for amphibian diseases (Grizzetti et al. 2011).

Like terrestrial eutrophication and N emissions to groundwater, aquatic eutrophication increases indirect N_2O emissions (see Chapter 3.3.3). Furthermore, eutrophication seems to enhance CH_4 emissions, especially in the form of ebullition, and amplify temperature dependency of CH_4 emissions. This could lead to a significant acceleration of global warming. The effect has been mainly shown for shallow lakes and ponds (DelSontro et al. 2016, West et al. 2016, Davidson et al. 2018) but also coastal areas (Borges et al. 2017). The processes governing the relationship between temperature, nutrient concentrations and CH_4 emissions are to date not fully understood though. Finally, eutrophication can become a self-reinforcing process, as high nutrient loads and oxygen deficiencies may cause the release of nutrients previously bound in the sediments.

Generally, marine waterbodies are considered N-limited, whereas rivers and lakes are considered P-limited. Although the opposite may be true in some cases (see Chapter 4.1), eutrophication assessments usually build on this principle. Impact estimations for freshwater eutrophication range between $2.28*10^{-11}$ and $3.4*10^{-9}$ potentially disappeared fraction of species (PDF) per t P applied to soil (Goedkoop et al. 2009, Verones et al. 2020). Endpoint impacts of marine eutrophication has only been quantified by Verones et al. (2020) as $3.75*10^{-13}$ PDF per t N applied to soil. Considering an average annual fertilisation rate of 11.2 million t N and 2.7 million t P (see Chapter 1.2.), current fertilising practices are responsible of an annual loss of 0.003-0.401% and 0.0004% of species in freshwater and marine habitats, respectively.

Like NO_3 pollution of groundwater, aquatic eutrophication in the EU is mainly monitored as concentrations in the receiving media, whereas there is no uniform dataset on agricultural N and P emissions. Currently 25% of EU surface waters (including rivers, lakes, transitional and coastal waters) are affected by agricultural pollution (European Environmental Agency 2018). Similarly, 23% of assessed European sea area is considered problematic in terms of eutrophication. Uncertainties are high though, as only 23% of sea area was covered by the assessment (EEA 2019). Rybaczewska-Błażejowska and Gierulski (2018) recently quantified midpoint eutrophication impacts of European agriculture with 1.17-1.27 Mt N_{eq} and 0.58-0.64 Mt P_{eq} in 2013. With a WTP of EUR 5-20 per kg N emitted (van Grinsven et al. 2013) associated costs amount to EUR 5.8-25.5 billion. Matthey and Bünger (2020) recommend EUR 20.8 per kg N and EUR 153.5 per kg P yielding EUR 113.4-124.7 billion in total if applied to whole Europe. Other than that, economic impacts of eutrophication are to date poorly quantified and restricted to specific case studies as the relation between nutrient emissions and caused damage are site specific and not fully understood (e.g. Pretty et al. 2003, Huang et al. 2010, Gourevitch et al. 2021).

3.3.5. Impacts on agricultural soil quality

Basically, effects of fertilisation on agricultural soil are similar to those of terrestrial eutrophication described in Chapter 3.3.2. However, due to the active management of agricultural soils they may manifest in different ways. For instance, N fertilisation increases soil organic matter which affects many physical, chemical, and biological properties of soils, including soil structure, water holding capacity, aeration, compaction, risk of erosion, biodiversity, nutrient availability, and the cation exchange capacity (CEC). Thus, soil high in organic matter can contribute to decreasing the necessary input of pesticides, labour, energy and capital and may improve crop genetic potential. Moreover, higher soil organic matter means more C sequestration in agricultural soil. However, fertiliser N has also been shown to promote the decomposition of crop residues and soil organic matter. Besides, other factors such as soil tillage, climate, and changes in land use probably have a larger effect on soil organic matter content than fertilisation (Butterbach-Bahl et al. 2011, Jensen et al. 2011, Velthof et al. 2011). Significant soil C sequestration has only been shown for manure fertilisation, although also here the extent is highly dependent on tillage intensity, climate, initial soil organic matter content, soil texture and pH, manure type, duration and intensity of application as well as on whether mineral fertiliser is applied simultaneously (Gross and Glaser 2021). In a global review study Gross and Glaser (2021) found increases in soil organic carbon content following manure application of 19-28% (95% confidence interval) in non-tropical climate with lowest values on sandy soil texture (15% on average) and highest for initial soil organic carbon contents of 1-2% (46% on average).

Most commonly used fertilisers in Europe are ammonium based and thus contribute to soil acidification. Manure also exhibits high concentrations of ammonium. Like in natural soils low pH increases the mobility of heavy metals in soil. On the one hand, agricultural soils may exhibit elevated concentrations of cadmium (Cd), zinc (Zn) and copper (Cu) due to long term inputs via (mineral and organic) fertilisers. On the other hand, liming can effectively compensate for acidification effects. However, liming constitutes an additional financial burden for farmers. Moreover, if agricultural soils are abandoned and liming is discontinued the risk of pollutant release to the environment prevails (Velthof et al. 2011).

As mentioned above fertilisers may contain pollutants that can accumulate in soil, crops and groundwater and might thus pose a risk to human and ecosystem health. Studies from north and west Europe indicate that the greatest risk to human health comes from cadmium (Cd), uranium (U), and nickel (Ni) present in mineral phosphate fertiliser as well as from copper (Cu) and zinc (Zn) which is contained both in mineral phosphate fertiliser and manure. While human exposure U, Ni, Cu, Zn is only through drinking water following leaching to groundwater, Cd is also taken up with food. Regarding soil organisms, Cd, Cu and Zn pose the highest environmental risk (Kraus et al. 2019, Pedersen et al. 2019). Table 1 shows the endpoint characterisation factors for damages to human health recommended by the EU (ILCD and PEF assessment methods, Fazio et al. 2018). For U to date no characterisation factor has been determined.

Table 1: Endpoint characterisation factors for human health of pollutants present in mineral and organic fertilisers as recommended by ILCD/PEF (Fazio et al. 2018). DALY: Disability adjusted life years.

Substance	Type	DALY/kg
Cd	Carcinogenic	0.00554
Cd	Non-carcinogenic	0.349
Cu	Non-carcinogenic	0.000101
Ni	Carcinogenic	0.00122
Ni	Non-carcinogenic	0.0000161
Zn	Non-carcinogenic	0.118

Based on Zn and Cu concentrations in manure (Leclerc and Laurent 2017) and amounts of manure applied to agricultural fields in the EU (Eurostat 2021a), annual damage related to organic fertilisation amounts to 9 billion DALYs or EUR 521-1254 billion, assuming EUR 57 700 – 138 700 per DALY (see Chapter 3.3.1). However, as most risk assessments show only limited exposure to Cu and Zn (Monteiro et al. 2010, Kraus et al. 2019, Pedersen et al. 2019), this is likely to be an overestimation. Regarding mineral fertiliser, mean Cd concentrations of mineral fertilisers are 32 mg/kg P_2O_5 (Smolders 2017). For an annual fertiliser application of 2.7 million t P_2O_5 (see Chapter 1.2.) this equals a release of 86.4 t Cd to agricultural soil, a loss of 30 632 healthy life years and corresponding costs of EUR 1.8-4.2 billion.

3.3.6. Impacts on land use change

Fertilisation increases crop yield so that more food can be produced on a smaller amount of land. Theoretically, this reduces the need for land use changes from forest or natural grasslands to cropland. Under European climate, fertilisation is estimated to increase yields by 40-60% (Stewart et al. 2005). For a fertilised area of 133.8 million ha (Fertilizer Europe 2020), thus, 53.5-80.3 million ha of native land, either within Europe or elsewhere in the world, could be preserved. De Groot et al. (2012) have reviewed global estimates of economic values of different ecosystems and found values between EUR 1161 per ha and year (woodlands) and EUR 3847 per ha and year (tropical forests) for terrestrial ecosystems. European fertilisation could thus contribute to the preservation of native land worth EUR 62.1-308.8 billion.

However, in practise, the relation between fertilisation and land use changes is less clear, as the conversion of native land to cropland is influenced by a multitude of factors including trade and market prices, economic development and national policies and regulations (Jensen et al. 2011).

3.3.7. Impacts on food quality

Like N deposition, which may change plant palatability for herbivores, N fertilisation can alter the quality of food and feed crops. While the positive correlation between N fertilisation and baking quality of wheat flour is well known, positive and negative effects of fertilisation on barley interact, making it difficult to determine an optimal level of N application. Feed crop quality may even be impaired by fertilisation, as high levels of N application may lead to a relative decline in essential amino acids such as lysine (Jensen et al. 2011).

3.3.8. Impacts on odour nuisance

Manure management and spreading of organic fertiliser can cause odour nuisance for residents in the vicinity. Odour is the second most frequent environmental complaint across Europe and 15% of odour sources stem from agriculture and livestock. Apart from being a nuisance that affects peoples' quality of life, it can have economic impacts (e.g. on property values or in the tourism sector) and is linked to several health issues (e.g. headaches, throat and eye irritation, nausea, sleeplessness, anxiety, stress, and respiratory problems). However, to date there is no common criteria to establish impact odour thresholds, making it difficult to quantify the nuisance (Rüfenacht et al. 2019).

3.4. Environmental impacts of mineral fertiliser production

3.4.1. Environmental impacts of N-fertiliser production

Irrespective of the fertiliser type, production of mineral N-fertilisers always starts with the conversion of airborne N_2 into reactive NH_3. This process is very energy intensive and globally accounts for approximately 2% of the world's energy demand (Sutton et al. 2013). Furthermore, the production of nitric acid (HNO_3), which is required for nitrate fertilisers, is a source of N_2O emissions. Almost half of N-fertilisers used in Europe are nitrate fertilisers (see Chapter 1.2). In addition, N-fertiliser production can cause emissions of CO_2, CH_4, NH_3, NO_x, SO_x, dust and particular matter to air and emissions N to water.

Table 2 gives an overview of the main environmental impacts of different types of N-fertilisers.

Table 2: Main environmental impacts of the main mineral nitrogen fertilisers used in Europe. Values for average European and global production as well as for best available technology (as of ca. 2010). AN: ammonium nitrate, CAN: Calcium ammonium nitrate, UAN: urea ammonium nitrate, AS: ammonium sulphate, n.a.: no data available.

	Nitrates (AN, CAN)	Urea	UAN	AS	References
Primary energy consumption [MJ/kg N]					
Europe	21.5 – 42.6	51.6	n.a.	42	Basosi et al. 2014, Skowrońska and Filipek 2014
Global	23.4 – 42.6	n.a.	n.a.	n.a.	Basosi et al. 2014
BAT	16.7 – 34.2	44.1	n.a.	n.a.	Ahlgren et al. 2009, Basosi et al. 2014, Skowrońska and Filipek 2014
Global warming potential [kg CO_2eq/kg N]					
Europe	3.3 – 6.3	0.9 – 4.0	1.3 – 6.2	3.0	Basosi et al. 2014, Skowrońska and Filipek 2014, Hoxha and Christensen 2019
Global	6.1 – 11.2	1.6 – 4.9	3.9 – 7.9	n.a.	Hoxha and Christensen 2019
BAT	2.3 – 2.8	0.9 – 1.13	n.a.	n.a.	Ahlgren et al. 2009, Basosi et al. 2014, Skowrońska and Filipek 2014,
Eutrophication potential [g O_2eq/kg N]					
Europe	34.4 – 37.9	37.2	n.a.	35.8	Skowrońska and Filipek 2014
Global	n.a.	n.a.	n.a.	n.a.	
BAT	30.0 – 31.0	n.a.	n.a.	n.a.	Ahlgren et al. 2009
Acidification potential [g SO_2eq/kg N]					
Europe	4.7 – 5.3	5.3	n.a.	5.3	Skowrońska and Filipek 2014
Global	n.a.	n.a.	n.a.	n.a.	
BAT	2.0 – 2.1	n.a.	n.a.	n.a.	Ahlgren et al. 2009

3.4.2. Environmental impacts of P-fertiliser production

The main environmental impacts of different types of P-fertilisers are shown in Table 3. Impacts are mainly associated with the use of sulphuric- and phosphoric acid (Kraus et al. 2019).

Table 3: Main environmental impacts of the main mineral phosphorus fertilisers used in Europe. Values for average European production (note that values from Kraus et al. 2019 are representative of production in Germany only) SSP: single superphosphate, TSP: triple superphosphate.

	Raw phosphate	SSP	TSP	References
Primary energy consumption [MJ/kg P_2O_5]				
Europe	12.0	7.3 – 30.0	16.9 – 27.0	Skowrońska and Filipek 2014, Kraus et al. 2019
Global warming potential [kg CO_2eq/kg P_2O_5]				
Europe	0.8	0.3 – 1.3	0.9 – 1.2	Skowrońska and Filipek 2014, Kraus et al. 2019
Eutrophication potential [g O_2eq/kg P_2O_5]				
Europe	6.0	21.9 – 78.0	28.4 – 78.0	Skowrońska and Filipek 2014, Kraus et al. 2019
Acidification potential [g SO_2eq/kg P_2O_5]				
Europe	5.0	3.7 – 17.0	4.5 – 17.0	Skowrońska and Filipek 2014, Kraus et al. 2019

In addition, during the production of phosphoric acid, 5 t of phosphogypsum are generated per tonne of P_2O_5 as a by-product. In lack of legal requirements for and economic viability of reuse, phosphogypsum is usually deposited in wet form in large stacks. As few of these stacks have functioning base seals or barriers against leachate losses, emissions of P, radioactive material and other pollutants occurs even long after the stacks have been closed.

3.4.3. Environmental impacts of K-fertiliser production

Environmental impacts of mineral K-fertilisers have been rarely assessed to date. Table 4 shows an example of the main environmental impacts for the production of muriate of potash (MOP).

Table 4: Main environmental impacts of the production of muriate of potash (MOP). Values for average European production.

	MOP	References
Primary energy consumption [MJ/kg P_2O_5]		
Europe	8.4	Skowrońska and Filipek 2014
Global warming potential [kg CO_2eq/kg P_2O_5]		
Europe	0.5	Skowrońska and Filipek 2014
Eutrophication potential [g O_2eq/kg P_2O_5]		
Europe	17.1	Skowrońska and Filipek 2014
Acidification potential [g SO_2eq/kg P_2O_5]		
Europe	6.0	Skowrońska and Filipek 2014

3.4.4. Environmental impacts of compound fertiliser production

The most commonly used mineral P-fertiliser in Europe is diammonium phosphate (DAP), a compound fertiliser containing 18% N and 46% P_2O_5 (Hasler et al. 2017). Furthermore, compound fertilisers account for 13% of mineral N-fertilisers used. Table 5 shows examples of compound fertilisers and their main environmental impacts.

Table 5: Main environmental impacts of compound fertilisers used in Europe (examples). Values refer to average production conditions in Germany. MAP: monoammonium phosphate, DAP: diammonium phosphate.

	MAP	DAP	NPK (15-15-15)	NP (20-20-0)	PK (0-12-20)	References
Primary energy consumption [MJ/kg fertiliser]						
Europe	21.8	23.9	15.9 – 16.6	18.8 – 52.4	6.2	Kraus et al. 2019
Global warming potential [kg CO_2eq/kg fertiliser]						
Europe	0.9	1.1	0.8 – 1.2	1.1 – 3.1	0.4	Kraus et al. 2019
Eutrophication potential [g O_2eq/kg fertiliser]						
Europe	14.6	12.9	13.0 – 14.4	20.0 – 41.6	11.0	Kraus et al. 2019
Acidification potential [g SO_2eq/kg fertiliser]						
Europe	13.5	13.8	8.0 – 8.3	10.6 – 27.0	3.2	Kraus et al. 2019

3.4.5. Monetising environmental impacts of fertiliser production

Environmental impacts of mineral fertiliser production can be assessed both from a consumption- and a production-based perspective. The former focuses on impacts of European fertiliser production, irrespective of the destination of the produced fertilisers (domestic agriculture, industry, export, etc.), whereas under the consumption-based approach impacts of production of fertilisers used in European agriculture (stemming both from domestic production and imports) are assessed. Table 6 provides an overview of the amounts and different types produced and consumed in Europe. Data stem from Fertilizer Europe (2020, see also Chapter 1.2). Shares of different types of P fertilisers in total consumption were taken from Kraus et al. (2019), assuming that values for Germany are representative of whole Europe.

Table 6: Amounts and types of fertilisers produced and consumed in the European Union (Fertilizer Europe 2020, Kraus et al. 2019). UAN: urea ammonium nitrate, SSP: single superphosphate, TSP: triple superphosphate, MAP: monoammonium phosphate, DAP: diammonium phosphate.

	Domestic production	Domestic agricultural consumption	Share of imports in domestic agricultural production
N-fertiliser [million t N]			
Total	13.4	11.3	28%
Nitrates		5.2	
Urea		2.4	
UAN		1.5	
Compound fertiliser		1.5	
Other		0.8	
P-fertilisers [million t P_2O_5]			
Total	2.0	2.7	66%
Raw phosphate		0.1	
SSP		0.0	
TSP		0.1	
MAP		0.1	
DAP		1.6	
NP (20-20-0)		0.2	
NPK (15-15-15)		0.4	
PK (0-12-20)		0.2	
K-fertiliser [million t K_2O]			
Total	2.7	3.1	71%

Following the approaches of monetising environmental impacts described in the previous chapters, the costs of 1 t CO_2 emissions can be estimated with EUR 31-805 (see Chapter 3.3.1). Eutrophication effects amount to EUR 5.0-20.8 per kg N and EUR 153.5 per kg P (see Chapter 3.3.4), which is equivalent to EUR 0.3-1.1 per kg O_2. (Ahlgren et al. 2009). Costs of acidification have been estimated by Matthey and Bünger (2020) for Germany with EUR 1500 per t SO_2 emitted. Regarding the impacts of leachate from phosphogypsum stacks, costs can be estimated with EUR 18.3 per kg P_2O_5, which corresponds to the abatement costs of installing a stack with functioning base seals and leaching barriers (Kraus et al. 2019).

Given these estimates, total environmental costs of mineral fertiliser production can be calculated as shown in Table 7 - Table 9. The following additional assumptions were made:

- For the production-based approach shares of different fertiliser types in domestic production were assumed equal to their share in agricultural consumption.
- For the consumption-based approach global average impacts were used for production of imported fertilisers, where available.
- As values for average impacts of European N-fertiliser production date back to approximately 2010 and technological improvements can be assumed, values for BAT were included in the impact ranges of European N-fertiliser production.
- The amount of compound fertiliser was upscaled from their P_2O_5 content. N content in compound fertiliser derived through this calculation corresponds approximately to the amount listed in Table 12 (1.3 million t N vs 1.5 million t N). 0.2 million t N difference were

added to other N fertilisers (i.e. 1.0 million t N instead of 0.8 million t N). Accordingly, K_2O content in compound fertilisers amounts to 0.8 million t K_2O.

- Impacts of other N fertiliser were assumed to be equal to impacts of ammonium sulphate (AS) and impacts of K_2O fertilisers not contained in compound fertilisers (2.3 t K_2O) equal to those of MOP.

Regarding leachate from phosphogypsum stacks costs amount to EUR 36.6 billion under the production-based approach and EUR 49.1 billion under the consumption-based approach.

Table 7: Global warming costs of mineral fertiliser production. Note that mass of single nutrient (SN) fertilisers are given in Mt of nutrient (N, P_2O_5, K), whereas mass of compound fertilisers refers to total mass. UAN: urea ammonium nitrate, RP: raw phosphate, SSP: single superphosphate, TSP: triple superphosphate, MOP, muriate of potassium, MAP: monoammonium phosphate, DAP: diammonium phosphate. For description of production- and consumption-based approaches see the main text.

		Production-based approach			Consumption-based approach				
	Unit costs [EUR/t CO_2eq]	Mass [Mt]	Impact [t CO_2eq/t]	Costs [mio EUR]	Mass from domestic production [t]	Impact domestic production [t CO_2eq/t]	Mass from imports [t]	Impact imports [t CO_2eq/t]	Costs [mio EUR]
N fertiliser (SN)	**31 - 805**	**12.4**		**749 – 53 201**	**7.9**		**3.4**		**925 – 53 578**
Nitrates	31 - 805	6.2	2.3 – 6.3	439 – 31 261	3.7	2.3 – 6.3	1.5	6.1 – 11.2	542 – 32 103
Urea	31 - 805	2.8	0.9 – 4.0	79 – 9061	1.7	0.9 – 4.0	0.7	1.6 – 4.9	81 – 8122
UAN	31 - 805	1.7	1.3 – 6.2	70 – 8694	1.1	1.3 – 6.2	0.4	3.9 – 7.9	92 – 7895
Other	31 - 805	1.7	3.0	161 – 4185	1.4	3.0	0.8	3.0	210 – 5458
P fertiliser (SN)	**31 - 805**	**0.1**		**4 – 143**	**0.0**		**0.1**		**5 – 193**
RP	31 - 805	0.0	0.8	1 – 26	0.0	0.8	0.0	0.8	1 – 35
SSP	31 - 805	0.0	0.3 – 1.3	0 – 21	0.0	0.3 – 1.3	0.0	0.3 – 1.3	0 – 28
TSP	31 - 805	0.1	0.9 – 1.2	3 – 97	0.0	0.9 – 1.2	0.1	0.9 – 1.2	4 – 130
MOP	**31 - 805**	**2.7**	**0.5**	**33 – 859**	**0.6**	**0.5**	**1.7**	**0.5**	**36 – 940**
Compound fertiliser	**31 - 805**	**6.9**		**187 – 6941**	**3.2**		**6.2**		**252 – 9370**
MAP	31 - 805	0.2	0.9	5 – 139	0.1	0.9	0.2	0.9	7 – 188
DAP	31 - 805	2.5	1.1	85 – 2195	1.1	1.1	2.2	1.1	114 – 2963
NP (20-20-0)	31 - 805	0.9	1.1 – 3.1	31 – 2246	0.4	1.1 – 3.1	0.8	1.1 – 3.1	41 – 3032
NPK (15-15-15)	31 - 805	2.0	0.8 – 1.2	50 – 1932	0.9	0.8 – 1.2	1.8	0.8 – 1.2	67 – 2608
PK (0-12-20)	31 - 805	1.3	0.4	17 – 429	0.6	0.4	1.2	0.4	22 – 580
Total				**973 – 61 144**					**1219 – 64 081**

Table 8: Eutrophication costs of mineral fertiliser production. Note that mass of single nutrient (SN) fertilisers are given in Mt of nutrient (N, P_2O_5, K), whereas mass of compound fertilisers refers to total mass. UAN: urea ammonium nitrate, RP: raw phosphate, SSP: single superphosphate, TSP: triple superphosphate, MOP, muriate of potassium, MAP: monoammonium phosphate, DAP: diammonium phosphate. For description of production- and consumption-based approaches see the main text.

		Production-based approach			Consumption-based approach				
	Unit costs [EUR/kg O_2eq]	Mass [Mt]	Impact [kg O_2eq/t]	Costs [mio EUR]	Mass from domestic production [t]	Impact domestic production [kg O_2eq/t]	Mass from imports [t]	Impact imports [kg O_2eq/t]	Costs [mio EUR]
N fertiliser (SN)	**0.3 – 1.1**	**12.4**		**125 – 512**	**7.9**		**3.4**		**463**
Nitrates	0.3 – 1.1	6.2	30 – 38	55 – 257	3.7	30 – 38	1.5	30 – 38	114 – 217
Urea	0.3 – 1.1	2.8	37	31 – 115	1.7	37	0.7	37	47 – 97
UAN	0.3 – 1.1	1.7	37	19 – 71	1.1	37	0.4	37	26 – 60
Other	0.3 – 1.1	1.7	36	19 – 68	1.4	36	0.8	36	16 – 89
P fertiliser (SN)	**0.3 – 1.1**	**0.1**		**1 – 11**	**0.0**		**0.1**		**1 – 14**
RP	0.3 – 1.1	0.0	6	0	0.0	6	0.0	6	0
SSP	0.3 – 1.1	0.0	22 – 78	0 – 2	0.0	22 – 78	0.0	22 – 78	0 – 2
TSP	0.3 – 1.1	0.1	28 – 78	1 – 9	0.0	28 – 78	0.1	28 – 78	1 – 12
MOP	**0.3 – 1.1**	**2.7**		**11 – 40**	**0.6**	**17**	**1.7**	**17**	**12 – 44**
Compound fertiliser	**0.3 – 1.1**	**6.9**	**17**	**28 – 127**	**3.2**		**6.2**		**38 – 172**
MAP	0.3 – 1.1	0.2	15	1 – 3	0.1	15	0.2	15	1 – 4
DAP	0.3 – 1.1	2.5	13	10 – 35	1.1	13	2.2	13	13 – 47
NP (20-20-0)	0.3 – 1.1	0.9	20 – 42	5 – 41	0.4	20 – 42	0.8	20 – 42	7 – 56
NPK (15-15-15)	0.3 – 1.1	2.0	13 – 14	8 – 32	0.9	13 – 14	1.8	13 – 14	11 – 43
PK (0-12-20)	0.3 – 1.1	1.3	11	4 – 16	0.6	11	1.2	11	6 – 22
Total				**165 – 690**					**153 – 649**

Table 9: Acidification costs of mineral fertiliser production. Note that mass of single nutrient (SN) fertilisers are given in Mt of nutrient (N, P_2O_5, K), whereas mass of compound fertilisers refers to total mass. UAN: urea ammonium nitrate, RP: raw phosphate, SSP: single superphosphate, TSP: triple superphosphate, MOP, muriate of potassium, MAP: monoammonium phosphate, DAP: diammonium phosphate. For description of production- and consumption-based approaches see the main text.

		Production-based approach			Consumption-based approach				
	Unit costs [EUR/kg SO_2eq]	Mass [Mt]	Impact [kg SO_2eq/t]	Costs [mio EUR]	Mass from domestic production [t]	Impact domestic production [kg SO_2eq/t]	Mass from imports [t]	Impact imports [kg SO_2eq/t]	Costs [mio EUR]
N fertiliser (SN)	**1.5**	**12.4**		**68 – 99**	**7.9**		**3.4**		**64 – 90**
Nitrates	1.5	6.2	2.0 – 5.3	18 – 49	3.7	2.0 – 5.3	1.5	2.0 – 5.3	16 – 41
Urea	1.5	2.8	5.3	22	1.7	5.3	0.7	5.3	19
UAN	1.5	1.7	5.3	14	1.1	5.3	0.4	5.3	12
Other	1.5	1.7	5.3	14	1.4	5.3	0.8	5.3	18
P fertiliser (SN)	**1.5**	**0.1**		**1 – 3**	**0.0**		**0.1**		**1 – 5**
RP	1.5	0.0	5.0	0	0.0	5.0	0.0	5.0	0
SSP	1.5	0.0	3.7 – 17	0 – 1	0.0	3.7 – 17	0.0	3.7 – 17	0 – 1
TSP	1.5	0.1	4.5 – 17	1 – 3	0.0	4.5 – 17	0.1	4.5 – 17	1 – 3
MOP	**1.5**	**2.7**	**6.0**	**19**	**0.6**	**6.0**	**1.7**	**6.0**	**9**
Compound fertiliser	**1.5**	**6.9**		**100 – 123**	**3.2**		**6.2**		**126 – 157**
MAP	1.5	0.2	13.5	4	0.1	13.5	0.2	13.5	5
DAP	1.5	2.5	13.8	51	1.1	13.8	2.2	13.8	69
NP (20-20-0)	1.5	0.9	10.6 – 27.0	14 – 36	0.4	10.6 – 27.0	0.8	10.6 – 27.0	19 – 49
NPK (15-15-15)	1.5	2.0	8.0 – 8.3	24 – 25	0.9	8.0 – 8.3	1.8	8.0 – 8.3	32 – 34
PK (0-12-20)	1.5	1.3	3.2	6	0.6	3.2	1.2	3.2	9
Total				**189 – 245**					**201 – 261**

3.5. Social and other impacts of mineral fertiliser production

3.5.1. Risk for major accidents associated with fertiliser production

In the EU control of major-accident hazards involving dangerous substances is regulated by the SEVESO Directive (2012/18/EU). Industrial establishments using or storing chemicals classified as dangerous above certain thresholds are obliged to deploy accident prevention policy and report accidents to the Major Accident Recording System of the European Commission (eMARS, European Commission 2020a). Since 2000 22 major accidents involving fertiliser production plants, production plants of ammonia which is at least partly used for fertiliser production and fertiliser storage facilities from EU member states, Norway, Iceland and Switzerland (the latter on a voluntary basis) have been recorded in eMARS (see Table 10). By far the largest incident was an explosion of ammonium nitrate at a fertiliser plant in Toulouse, France on September 21, 2001. 31 people were killed and 2772 injured, 30 of which seriously. Material damage amounted to more than EUR 2 billion and use of tap water was temporarily prohibited due to pollution. In addition, during emergency response an uncontrolled release of 9 t ammonia solution to the Garonne River occurred, causing fish death. Fatalities were recorded in three other cases: In 2007, a worker of a manufacturing plant of chemical products for agricultural, construction, chemical processing and plastic industries died from carbon monoxide intoxication after uncontrolled release of synthesis gas, another worker of a fertiliser production plant was killed by uncontrolled release of ammonia in 2014 and in the same year a worker succumbed to his injuries after being splashed and burned by reaction solution for the production of magnesium nitrate. Of the remaining incidents, four involved the explosion of ammonium nitrate, ten explosion and/or ignition of other substances, seven uncontrolled release of ammonia and eleven uncontrolled releases of other toxic or environmentally hazardous substances. In total 140 people were injured, and material damage amounted to more than EUR 46 million (excluding the Toulouse accident).

Table 10: Major accidents under the SEVESO directive involving fertiliser production plants, production plants of ammonia which is at least partly used for fertiliser production and fertiliser storage facilities recorded in the eMARS database (European Commission 2020) since 2000. As reporting processes following an accident usually take two to three years, records of the final three years of data may not be complete. Note that there is no common definition of serious, medium and minor injuries between the accidents.

Date	Site	Accident type	Substance involved	Cause	Deaths	Injuries	Material damage	Environmental damage	Other disruption
21.09.2001	N-fertiliser production plant	Explosion Fire Release of combustion products Domino effects	AN: 20 t	Other	31 On-site: 22 Off-site: 9	2772 Serious: 30 Medium: 300 Minor: 2442	BEUR 2.01 On-site: BEUR 2 Off-site: MEUR 10	On-site soil pollution: MEUR 100 Pollution of local river Fish death	4h confinement of city Interruption of traffic Interruption of communication lines within a 100 km radius 7 day-ban on consuming tap water
26.01.2002	Fertiliser production plant	Fire Release of combustion products	NPK (15:15:15): 15 372 t NH_3 N_2O	Inadequate management Inadequate procedures Unexpected reaction	0	30 Serious/medium: 1 Minor: 29	unspecified	unspecified	Evacuation of houses and installations in proximity 3 day disruption of infrastructure
12.10.2002	Oil refining industry	Explosion Fire	AN: 15 372 t Kerosine: 800 000 t Gas oil: 70 m³ H_2: 200 Nm³	Inadequate management Inadequate procedures Inadequate design Pipe failure Unexpected reaction	0	0	unspecified	unspecified	Confinement of neighbouring municipality Traffic interruption
28.05.2003	Ammonia plant	Fluid release to water Release to air	As: 0.75 t	Technical failure	0	0	0	River pollution	unspecified

02.10.2003	Tree plantation warehouse	Explosion Fire Release of combustion products	AN: 3-5 t	Other	0	23 Serious: 9 Medium/minor: 14	unspecified	unspecified	4 day-evacuation of 60 residents
01.06.2006	N-fertiliser production plant	Explosion Fire Release of combustion products	Synthesis gas (75% H_2, 20% NH_3, 5% other)	Valve failure	0	2 Serious: 0 Medium: 0 Minor: 2	On-site: MEUR 2	unspecified	6 week closure of plant
12.02.2007	N-fertiliser production plant	Fluid release to water	Aminoguanidine formate: 5.6 t Aminotriazole: 0.007 t Hydrazine hydrate: 0.13 t	Inadequate management Inadequate procedures Inadequate design	0	0	0	Freshwater pollution	unspecified
27.04.2007	Urea production plant	Release to air	NH_3: 6 t	Plant/equipment failure	0	unspecified Serious: 0 Medium: 0 Minor: unspecified	0	unspecified	Temporary closure of plant
23.08.2007	Chemical plant	Release to air	CO	Inadequate procedures Lack of training/supervision Monitoring device failure	1 On-site: 1 Off-site: 0	0	0	none	unspecified
10.09.2007	NPK fertiliser production plant	Release to air	Urea NH_3	Inadequate procedures Human error	0	9 Serious: 0 Medium: 1 Minor: 8	0	unspecified	unspecified

03.12.2008	Fertiliser production plant	Explosion Fire	AN	Inadequate process analysis Inadequate plant design Loss of process control Equipment failure Human error	0	5 Serious: 0 Medium: 0 Minor: 5	On-site: MEUR 2	unspecified	Production loss: 150 000 t NPK, 60 000 t CAN
26.06.2009	Ammonia plant	Explosion Fire	Natural gas (CH_4, H_2)	Inappropriate inspecting Loss of process control Human error Natural event Utilities failure	0	2 Serious: unspecified Medium: unspecified Minor: unspecified	unspecified On-site: unspecified Off-site: 0	unspecified	unspecified
13.08.2009	Fertiliser production plant	Release to air	NH_3: 0.2 t	Maintenance/repair Inadequate inspection Human error	0	24 Serious: 0 Medium: 4 Minor: 20	0	unspecified	unspecified
24.07.2010	Fertiliser production plant	Explosion Fire Release of combustion products	H_2: < 2.5 t CH_4: < 10 t	Equipment failure	0	5 Serious: unspecified Medium: unspecified Minor: unspecified	On-site: MEUR 12	none	unspecified
29.07.2012	Fertiliser production plant	Release to air	NH_3: 1.7 t	Inadequate procedure Inadequate management Equipment failure	0	6 Serious/medium: 6 Minor: 0	0	unspecified	unspecified

25.09.2012	NPK fertiliser production plant	Fire Release of combustion products	unspecified	Unknown	0	25 Serious: unspecified Medium: unspecified Minor: unspecified	Unspecified On-site: unspecified Off-site: unspecified	unspecified	Temporary confinement of neighbouring residents Temporary interruption of traffic
26.04.2013	KN fertiliser production and distribution plant	Release to air Fluid release to ground	Nitric acid: 1.3 t	Lack of training/supervision Human error	0	7 Serious: 0 Medium: 7 Minor: 0	On-site: several 100 EUR	none	Production interruption for 12 h
14.01.2014	Ammonia production plant	Fire Release of combustion products	H_2: 0.7 t	Inadequate inspection Pipe failure	0	0	On-site: MEUR 9	unspecified	Plant closure for several months
21.02.2014	Fertiliser production plant	Release to air	NH_3: 15 t	Inadequate procedures Lack of training/supervision Human error	1	0	0	unspecified	unspecified
27.07.2014	Chemical production plant	Fire Explosion Release to air	Nitric acid	Lack of supervision Human error	1	0	0	unspecified	unspecified
31.07.2016	Biogas plant	Fire Release of combustion products	Biogas: 3.95 t Sulphuric acid: 6.68 t	Unexpected reaction	0	2 Serious: 0 Medium: 0 Minor: 2	unspecified On-site: unspecified Off-site: 0	Release of 3000 m^3 biogas	unspecified
24.04.2017	Ammonia plant	Fluid release to ground Fluid release to water Fire Explosion	Syngas (0.5-2% H_2)	Inadequate procedures Lack of training Inadequate process analysis Other	0	0	On-site: MEUR 17.7	Limited oil release to the fjord, cleaned immediately	4 months plant closure

Also on a global scale, disastrous accidents in the fertiliser industry are often associated with explosion of ammonium nitrate. Although not flammable on its own and only able to explode by itself if rapidly heated to 240°C, ammonium nitrate acts as an oxidant during fire and can thus promote explosion (Gibbens 2020). Hence, accidents with ammonium nitrate are often caused by improper storage or handling. Of the ten largest accidental explosions in history, four involved ammonium nitrate. Since 2000 11 incidents have been recorded, of which five occurred in the EU (see Table 11).

Table 11: Explosions at fertiliser plants, storage sites or during transport of ammonium nitrates since 2000. Incidents in the EU are highlighted in bold (Ang 2020).

Year	Location	AN (tonnes)	Deaths
2001	**Toulouse, France**	**20**	**30**
2003	**Saint-Romain-en-Jarez, France**	**4.5**	**26**
2004	**Barracas, Spain**	**25**	**2**
2004	Ryongchŏn, North Korea	not reported	160
2004	**Mihăileşti, Romania**	**20**	**18**
2007	**Estaca de Bares, Spain**	**400**	**0**
2007	Monclova, Mexico	22	57
2013	West, Texas (USA)	27	15
2014	Wyandra, Australia	56	0
2015	Tianjin, China	726	165
2020	Beirut, Lebanon	2495	211

Major accidents during the production of P fertilisers are mainly associated with process water accumulating on the plateau of phosphogypsum stacks or in adjacent lagoons. In addition to continuous emissions (see 3.4.2) large amounts of P, radioactive substances and other pollutants can be released into the environment during dam failures or subsiding of the ground due to dissolution of minerals (so-called sinkholes). For instance, a dam failure caused by a storm at the phosphogypsum stack in Huelva, Spain on 31.12.1998, led to emissions of 50 000 – 500 000 m³ of acid process water to the marshes of the Tinto River. In Florida, sinkhole accidents releasing more than 100 000 m³ of acid process water to the environment have occurred in 1995, 1997, 2004 and 2016 (Kraus et al. 2019). The most recent incidents were a dam break in Mishor Rotem, Israel in July 2017 and a threatened pond collapse in Piney Point, Florida in April 2021 (Kraus et al. 2019, Gabbat 2021). In the Israelian case, around 100 000 m³ of high acid fluid were washed down a river valley towards the Dead Sea causing closure of a highway and death of animals that used the water points remaining in the riverbed during summer (Kraus et al. 2019, Sones 2017). In Piney Point, a breach of the pond walls and spill of over 2 billion m³ of wastewater could be prevented, however, more than 300 homes had to be evacuated and 814 million m³ nutrient-loaded water were pumped into the ecologically sensitive Tampa Bay. In the following summer, the region experienced a particularly severe event of red tide, toxic algal blooms that can cause respiratory problems for people, kill fish and other marine life, and cause shellfish poisoning in people, which may have been aggravated by the incident at Piney Point (Gabbat 2021, Gammon 2021). In addition to the immediate damage, plant operators are often allowed to use already decommissioned stacks with less effective barriers against leachate and safety requirements in the initial phase after the accident (Kraus et al. 2019). Restoration after large dam failures or sinkhole event may take 10 years.

However, to our knowledge damages have not been systematically assessed for any of the recent accidents at phosphogypsum stacks. Moreover, most of the incidents have been associated with improper deposition practice and poor maintenance of the stacks. Similarly, there are high differences between the incidents recorded in the eMARS database regarding the degree and detail to which injuries and material damage is reported and environmental damage is rarely quantified. At present it is therefore not possible to provide a statistical evaluation or monetary quantification for the risk of major accidents during production of mineral fertilisers. Nevertheless, from Table 10 it is clear that even if safety regulations are in place, factors such as human error, lack of implementation, unforeseen process reactions and unfortunate combination of rare events make it impossible to completely prevent the occurrence of disastrous events.

3.5.2. Labour quality and safety in the fertiliser industry

The European chemical industry is characterized by a comparatively high share of large workplaces (26% with more than 250 employees in 2010), above average wages (see Chapter 3.1), low self-employment levels (4% in 2010) and dominance of typical and regular working hours (Eurofound 2012). It is a male dominated sector with only 36% of female workers in 2010 and an even lower proportion of women in leading positions (16% compared to 28% on EU average).

In 2010, more than 90% of employees had an indefinite contract, compared to 80% in the EU (Eurofound 2012). Consequently, turnover is low, i.e., workers stay for a relatively long period in the same company (Eurofound 2009). Atypical work hours (weekends, evenings, nights) are almost 60% less common for women and more than 30% less common for men than on EU average, whereas regular work hours (same hours in a day, same days in a week) are more common (Eurofound 2012). Furthermore, the sector has a low part-time employment rate of 8% in 2016, compared to 21% on EU average (Eurostat 2021bf). However, in 2010 the proportion of employees who would prefer to work fewer hours was slightly higher than on EU average (35% vs 31%). Especially male employees in companies with more than 250 employees reported a poor work-life-balance (Eurofound 2012).

Like on EU average, more than half of workers in the chemical industry feel that their skills correspond well to their tasks and more workers consider themselves over-skilled than under-skilled. However, the proportion of workers considering themselves under-skilled is slightly higher than on EU average. As employer paid training is also more frequent than in other sectors this is probably a result of the high level of skills required in the sector. The comparatively high level of education is also the reason why wages and job prospects in the chemical industry are considered above average (Eurofound 2012).

Figure 9 shows the incidence rate for fatal and non-fatal accidents in the chemical industry over the last decade. While fatal accidents occur more often than on EU average, other accidents are less frequent and have been declining more pronouncedly since 2010. The incident rate for fatalities is well in line with the data for accidents in the fertiliser industry (see Table 10) where the average incident rate since 2010 is 2.5. The incident rate for non-fatal accidents is below 20, however, it has to be noted that only a fraction of accidents is recorded in the SEVESO database. Apart from work-related accidents, levels of poor self-reported health in 2010 were lower for the chemical industry and more workers felt able to continue doing their job at the age of 60. On the other hand, mental well-being was slightly lower and issues of absenteeism 5% higher than on EU average. Especially men below 35 years often find themselves in positions with a high level of demand but a low level of control over the way in which they carry out their task and are thus more likely to suffer from work-related stress.

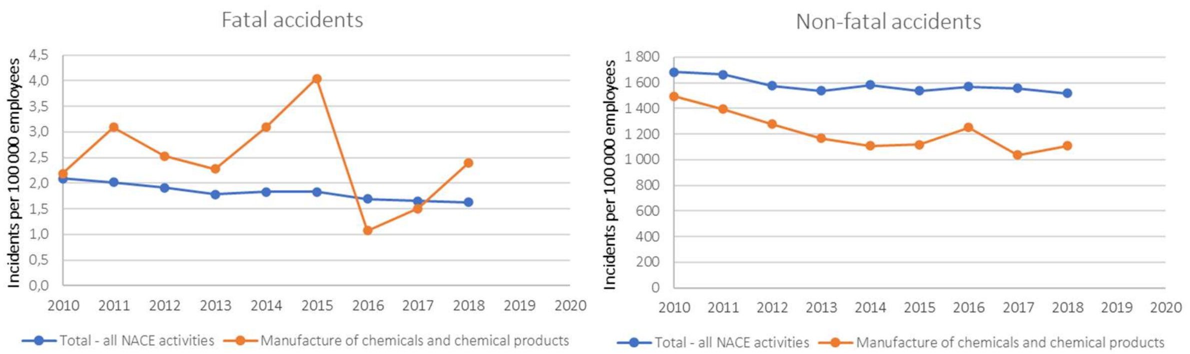

Figure 9: Incident rate of fatal and none-fatal accidents in the manufacturing sector of chemicals and chemical products and in all NACE activities in the EU since 2010.

For 75% of employees in the chemical industry employee representatives are available, whereas this is only the case for 51% of employees on EU average. The social work environment is also rated slightly better in the chemical industry (Eurofound 2012).

Regarding labour quality and safety along the supply chain, phosphate mines are of particular importance, as domestic mining is limited to one site in Finland. P_2O_5 is mainly imported from Morocco (28%), Russia (23%), Algeria (13%), Israel (8%) and Syria (7%) (European Commission 2020b). Health risks for mine workers include lung cancer due to exposition to radon as well as cardiovascular and respiratory problems and higher mortality due to exposition to particulate matter. Recent studies indicate that health impairments should be low if plants are equipped with filter systems according to the state of the art and if these systems are properly maintained (Kraus et al. 2019). However, the extent to which this is the case for the mines from which the EU sources phosphates remains unclear. Likewise, companies operating phosphate mines in main EU source countries report engagement in safe workplace policy, employee training, social support programmes and gender equality (e.g. Acron n.d., OCP 2020, PhosAgro 2021), but as different indicators are used and mine workers are not distinguished from other employees, no general statements can be made.

3.6. Social and other impacts of fertiliser use

3.6.1. Contribution of fertiliser use to food security

The most obvious benefit of mineral fertiliser use is undoubtedly its enabling of higher yields and consequent contribution to food security of a growing population. It is estimated that 27% of the world's population growth over the past century would not have been possible without mineral N fertiliser and that in 2008 N fertiliser fed 48% of the population (Erisman et al. 2008). Through yield increases in fodder plants N fertilisation also enabled an expansion in livestock numbers. Thus, mineral fertilisers are also indirectly responsible for the increase in availability of organic fertiliser in the form of manure (Roser and Ritchie 2013). Considering the value of a life year of EUR 57 700 – 138 700 (Holland 2012, see also Chapter 3.3.1) and an average population in the EU28 of 512 million in 2017-2019 (Eurostat 2022), monetised benefits of mineral fertiliser use in terms of fed population can be estimated with EUR 14 188 – 34 106 billion.

However, the global food system is complex and geographically highly uneven. For instance, long term experiments showed fertiliser induced yield increases of 40-60% in the USA and England, but up to 80-90% in the tropics (Stewart et al. 2005). Moreover, differences in affluence have led to high differences in the abundance of food. While fertiliser and food, especially meat, are often consumed in excess in the EU and other prosperous countries, with all the negative environmental consequences entailed, 8-10% of the global population remain undernourished. Thus, the share of population in high-income countries, which are dependent on mineral fertilisers to meet basic food needs, is likely considerably below 48%, whereas it may be higher in low-income countries (Roser and Ritchie 2013). Furthermore, as can be seen in Figure 10, wealthier countries invest more in mineral fertiliser use. The correlation between mineral fertiliser use and prevalence of undernourishment and obesity are much weaker or even insignificant though. This may on the one hand be due to the complex interconnections of the global food system, where food and feed are often not consumed where they are produced. On the other hand, food insecurity and famine are often rather a problem of stability, socio-political factors and economy than of inability to produce enough food (Roser and Ritchie 2013).

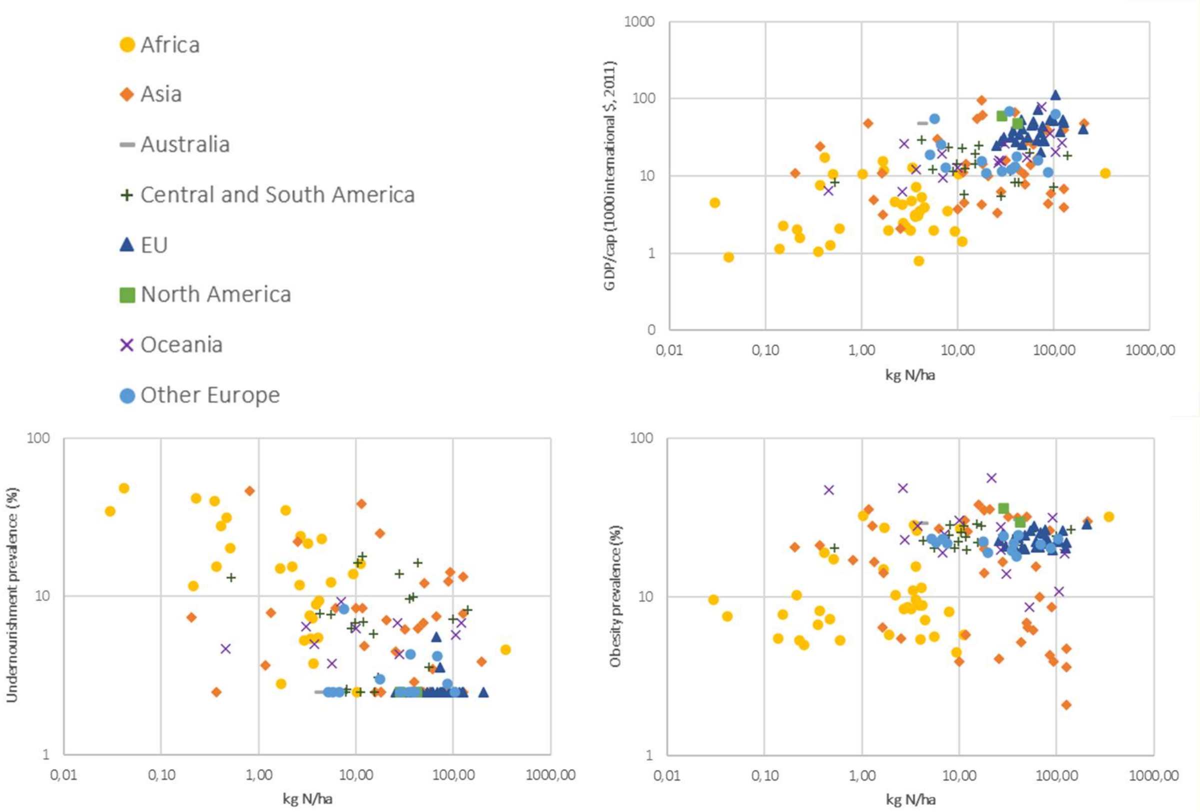

Figure 10: Correlation between mineral N fertiliser application, GDP, undernourishment and obesity in 2016. Note that both x and y axis of the graphs are shown in a logarithmic scale. Undernourishment prevalence below 2.5% of a country's population is not recorded and therefore shown as 2.5% in the graph (FAO 2021a,b).

3.6.2. Geopolitical impacts

Both P and K reserves are concentrated in few countries. 14 countries hold 98% of P reserves, of which 73% are located in Morocco and Western Sahara (European Commission 2020b). Similarly, Russia and Canada account for 80% of K reserves (Sutton et al. 2013). This not only poses a risk to future supply but may also affect the political and economic relations Europe entails to these countries. For instance, the EU's inconsistent position on the status of Western Sahara is partly due to its dependence on Moroccan phosphate imports. On the other hand, phosphate export is also an important economic factor for Morocco and the good trade relations could form a communication basis for finding a solution to the conflict (Cavanagh 2021, Rosemarin 2004). Similarly, plans to restrict Cd levels in phosphate fertilisers in the EU have raised concerns about increased economic and consequently political dependence on Russia, as Cd levels in Russian rock phosphate, for geological reasons, are considerably lower than in North African ones (Wanat 2020).

4. CASE STUDIES: EUTROPHICATION OF THE BALTIC SEA

4.1. Legacy impacts of over-fertilisation in terms of eutrophication of the Baltic Sea

4.1.1. Case study region description

Characteristics of the Baltic Sea and its catchment area

With 240 000 km² the Baltic Sea is one of the world's largest brackish water bodies. The characteristic salinity gradients both with depth and from the southwest to the northeast stem from the facts that the Baltic Sea is rather shallow (less than 30 m in more than one third of the area) and that water exchange is limited as the narrow passage through the Sound and Belt Sea is its only connection to the North Sea. The latter also causes seasonal oxygen deficiencies and anoxic conditions in deeper parts of the sea as well as accumulation of nutrients and other pollutants from human origins. These factors make the Baltic Sea a unique but vulnerable environment. (HELCOM 2018b).

The catchment area of the Baltic Sea comprises 1 729 500 km² in 14 different states. The coastal states Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Russia, and Sweden make up 93% of the catchment area; the remaining 7% belong to Belarus, Czech Republic, Norway, Slovakia, and Ukraine (Räike et al. 2019). Climatological conditions vary considerably across the basin with atlantic-temperate climate in the southwest, continental-temperate climate in the east and boreal to artic conditions in the north. This is reflected both in population density and land use, which tend to decrease from south to north (Figure 11a). Livestock densities and consequently manure application rates in the different regions are also very variable, although they are less correlated with geographic and climatological conditions (Figure 11b). Leningrad region with 182 kg/ha in 2015 exhibits one of the highest N inputs with manure in whole Europe (Kuka et al. 2019, Eurostat 2021a). P input with manure is highest in the German regions (17 kg/ha in 2014), whereas lowest rates are found in the Baltic states (Estonia: 13 kg N/ha, Latvia: 3 kg P/ha in 2014)[1] (Räike et al. 2019, Eurostat 2021a). Hotspots for P and N input with mineral fertiliser correlate percentage of agricultural land (see Figure 11c and Figure 11d). In 2016, application rates of mineral N fertiliser in the Baltic Sea catchment area were with 12 kg/ha lowest in Russia[2] (Our World in Data n.d.a) and with 181 kg/ha highest in Schleswig-Holstein (DE). For mineral P application, the range goes from 1 kg/ha in Övre Norrland (SE) to 15 kg/ha in Opolskie (PL).

[1] Except for Leningrad region, data on manure application rates is only available on national level, respectively the parts of a country lying within the Baltic Sea catchment.

[2] For Russia, Ukraine and Belarus application rates of mineral fertiliser are only available on national level.

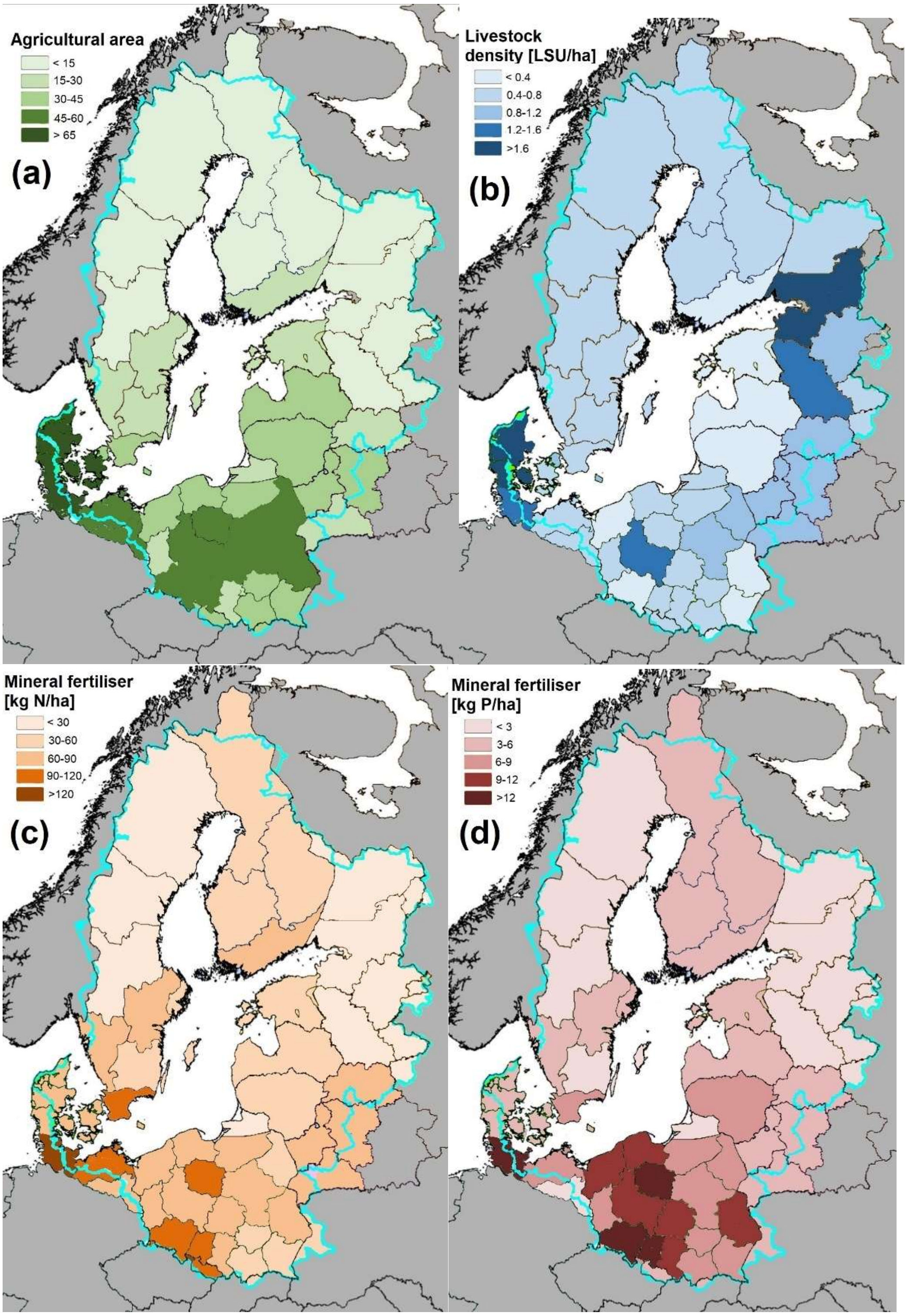

Figure 11: Fertilising practices in the Baltic Sea catchment. (a) Proportion of agricultural area of total land area in 2016 (Eurostat 2021f, Knoema n.d.a-h, Belstat 2020). No data available for St. Petersburg, Ukraine and Norway. (b) Livestock densities in 2016 (Eurostat 2020d, Knoema n.d.a-h, Belstat 2020). Data for Belarus only available on national basis. No data available for St. Petersburg, Ukraine, and Norway. Application of mineral N (a) and P (b) fertiliser in 2016 (Eurostat 2021b, Our World in Data n.d.a-b). Data for Belarus, Russia and Ukraine are only available on national level. Background maps: Svanbäck et al. (2019).

Efficiency and sustainability of current fertilising practices can be measured by the gross nutrient balances (GNB). The GNB is the difference between the nutrient inputs to agricultural land and outputs via crop harvest and -residues. Thus, it quantifies the amount of nutrients that are prone to leaching, run-off, erosion and, in case of N, atmospheric emissions (Eurostat 2013). Whereas the highest N surplus in 2015 occurred in Leningrad region (168 kg N/ha)[3] (Kuka et al. 2019), P surpluses were highest in Denmark (7 kg P/ha) and Finland (4 kg P/ha). The Baltic states on the other hand exhibit particularly low nutrient balances, with Estonia showing even a P deficit of 7 kg P/ha, meaning that more P is extracted with the harvest than is supplied by fertilisation (Eurostat 2021a). This can be a strategy to mitigate the effects of past overfertilisation and soil accumulation of P (Ylivainio et al. 2014), however, if P deficits prevail over a longer period, they will eventually lead to a decrease in soil fertility. In general, nutrient surpluses in the catchment decreased over the past years following measures taken to prevent nutrient losses and to increase fertiliser efficiencies. However, N surpluses are stagnating in Poland and have even increased over the past 20 years (although on a very low level) in Latvia (Eurostat 2021a).

Current eutrophication in the Baltic Sea

Based on an integrated assessment by HELCOM (Helsinki Commission), an intergovernmental alliance between the nine coastal Baltic countries and the EU, in the period 2011-2016 97% of the Baltic Sea were affected by eutrophication (HELCOM 2018c). Only small areas along the coasts of Sweden, Finland and Denmark were assigned good eutrophication status (Figure 12). The HELCOM assessment combines indicators on nutrient levels, direct effects (concentrations of chlorophyll-a, water quality as well as biomass, extent and intensity of cyanobacterial blooms) and indirect effects (oxygen debt and state of the soft-bottom macrofauna community) and is well in line with both reporting under the EU Water Framework Directive (WFD; 2000/60/EC) and a recent assessment by the European Environment Agency (EEA; EEA 2019). According to WFD reporting 58% of transitional and coastal waters of the Baltic Sea are below good status in terms of nutrients, while 66% of transitional and 85% of coastal waters exhibit deficient phytoplankton conditions. The EEA even designates 99.4% of the Baltic Sea as "problem area" making it the marine water body most severely affected by eutrophication within Europe.

Although eutrophication is partly a natural phenomenon and subject to natural variabilities of, among others, high inflow events during winter storms, human activities are responsible for the majority of nutrient load to the Baltic Sea. Figure 13 shows the development of nutrient inputs over the past century. The Baltic Sea has not been in a pristine status for 150 years and first signs of eutrophication already emerged 100 years ago (EEA 2019). However, from the 1950s on a steep increase in nitrogen and phosphorus emissions occurred, which is mainly associated with the intensification of agriculture and increased fertiliser use (EEA 2017). Since the 1980s, when the importance of reducing nutrient loads to maintain a healthy ecosystem was acknowledged, nutrient inputs have been reduced significantly. Apart from EU policies such as the WFD (2000/60/EC), the Marine Strategy Framework Directive (2008/56/EC), the Urban Wastewater Treatment Directive (91/271/EEC), the Nitrates Directive (91/676/EEC) and the Industrial Emissions Directive (2010/75/EU), the HELCOM Baltic Sea Action Plan (BSAP) is a key policy in achieving good eutrophication status in the region. The BSAP sets maximum allowable nutrient input levels (792 209 t of N and 21 716 t of P); see green lines in Figure 13) that should be achieved by 2021 (HELCOM 2018b). These targets are broken down both to sub-

[3] Data for Russia is only available for Leningrad region. For other countries, data on GNB is only available on national level. No data is available for Belarus and Ukraine.

basins and to reduction needs compared to reference inputs in 1997-2003 for individual member countries.

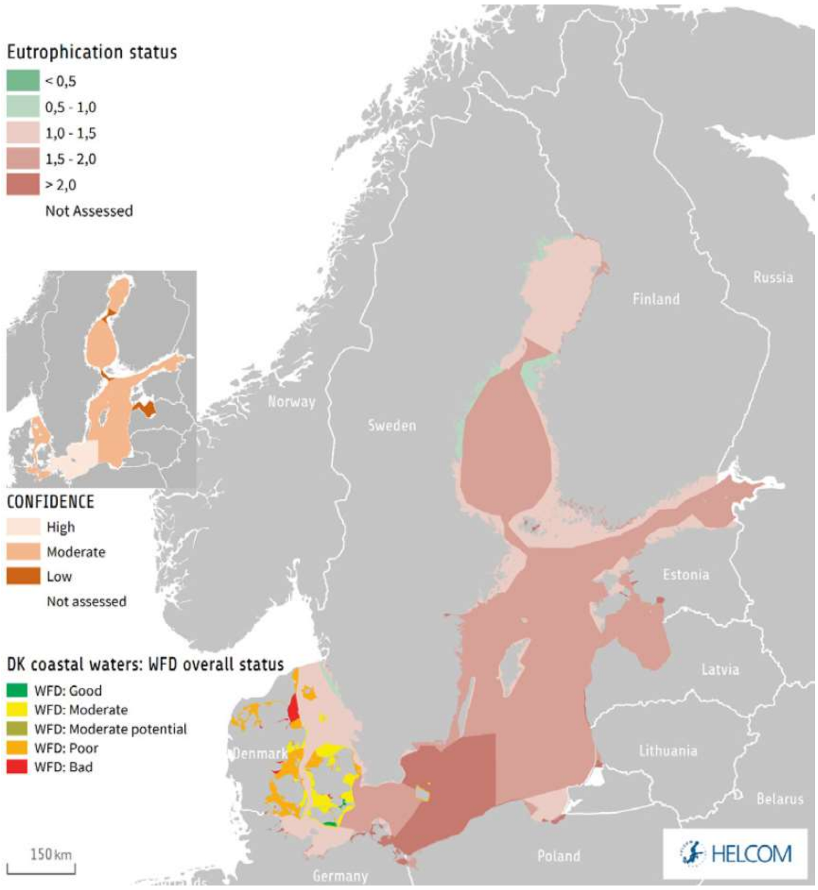

Figure 12: Integrated status of eutrophication in the Baltic Sea 2011-2016. Each assessment unit shows the result for the indicator group furthest away from good status. Numbers indicate the ratio between observed indicator values and pre-defined threshold values, where values >1 signify that the threshold value is not exceeded. Denmark assesses coastal waters according to WFD-classification; hence, colours are not directly comparable. Furthermore, eutrophication status in Kattegat and Great Belt may have been overestimated due to an underestimation of total nitrogen content in Danish marine and freshwater samples caused by an imprecise measurement method (HELCOM 2018c).

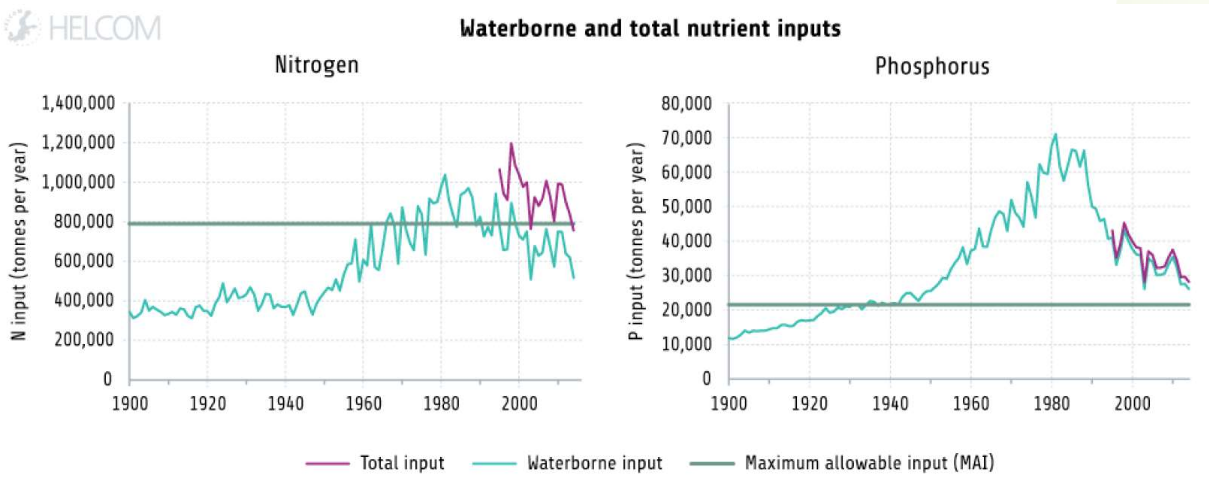

Figure 13: Temporal development of waterborne and total nutrient inputs to the Baltic Sea from 1900-2014. The green line shows the maximum allowable input (MAI) agreed on in the Baltic Sea Action Plan (HELCOM 2018c).

Nevertheless, in 2015, nutrient loads to the Baltic Sea amounted to 827 773 t of N and 30 026 t of P, meaning that targets of the BSAP were still exceeded by 7% for N and 44% for P. While direct discharges of municipal or industrial wastewater treatment facilities N and P emissions decreased by 50% and 67% respectively and currently account for less than 5% of total nutrient loads, reduction of diffuse riverine and airborne emissions has proven more challenging (HELCOM 2018d,e). In addition to lack of implementation (HELCOM 2018f), the effectiveness of the BSAP measures as such has been criticised. The WWF for instance claimed that some national programmes on nutrient reductions are too weak to reach the agreed targets and that there is an overvaluation of supporting actions like guideline development, monitoring, research and survey instead of concrete emission reductions (WWF 2018).

Even if the reductions agreed upon were achieved, the overarching aim “a Baltic Sea unaffected by eutrophication” (HELCOM 2007) would remain a considerable way off. On the one hand, the targets in the BSAP were set under the assumption of a steady state of the Baltic Sea environment (HELCOM 2018b) not taking into account global trends such as a growing and increasingly affluent global population, rising traffic on the Baltic Sea and climate change that point towards an increase in pressures to the marine environment and may therefore also have negative feedbacks on eutrophication (BalticSTERN Secretariat 2013). Climate change, in particular, through an increase in extreme precipitation events amplifying nutrient erosion from soil and riverbanks, shorter periods of soil frost and snow cover which protect nutrient from leaching and a potential expansion of agricultural area to the north following the warmer climate could undermine nutrient emission reduction efforts (Räike et al. 2019, BalticSTERN Secretariat 2013). In fact, taking climate change into account, the emission reductions agreed upon in the BSAP might just be enough to sustain the current eutrophication status, but not bring upon any improvements (EEA 2017).

On the other hand, legacy of past P emissions may delay the effect of emission reductions significantly. During the past century, huge amounts of P emitted to the Baltic Sea have been bound by different substances in the bottom sediments such as iron, aluminium, or calcium. Under anoxic conditions, some of these substances, predominately iron, loose their P-binding capacity and P is released to the water column. Moreover, P from past overfertilisation has been stored in agricultural soils or retained in terrestrial surface waters. Nitrogen is stored in the sediments to a much lesser degree and significant parts of N leave the Baltic Sea ecosystem via denitrification. On the other hand, in addition to the external load, N enters the Baltic Sea via N_2 fixation by cyanobacteria, whose growth is enhanced by higher levels of P in the water column. Thus, without tackling the P load, efforts to reduce N loads would be pointless. It is estimated that if legacy nutrient loads are not addressed it would take another 150-200 years until good ecological status in the Baltic Sea is reached (HELCOM 2018c, EEA 2019).

Both legacy P emissions and foreseen climate change impacts will play an important role in the current updating of the BSAP (HELCOM 2018b).

4.1.2. Objectives of the present case study

While being the marine water body most severely affected by eutrophication, standard environmental assessment methods typically do not apply to eutrophication in the Baltic Sea because it is limited by P rather than N (see also Chapter 3.3.4). In addition, the fact that economic conditions, population densities and agricultural practices exhibit a large variability throughout the catchment and thus cover a large part of the European spectrum make the area an interesting hotspot region to study.

The Baltic Sea is exceptionally well studied and monitored (EEA 2019) and over the past 20 years several studies have quantified the costs of its eutrophication as economic damage (e.g. Dahlgren et al. 2015), WTP (e.g. Ahtiainen et al. 2014), abatement costs (e.g. Gren 2008, Hasler et al. 2012, BalticSTERN Secretariat 2013, Ahlvik et al. 2014, Hasler et al. 2014) or with stated preference (e.g. Czajkowski et al. 2015). Yet, these have mainly focussed on reducing current nutrient emissions, while the costs of legacy P have not been systematically analysed to date. This is therefore the issue of the present case study. Results have also been published in Tanzer et al. (2021).

4.1.3. Quantifying agricultural legacy nutrient loads

Nutrients present in the sea

As described in Tanzer et al. (2021), the share of current nutrient pools attributable to agricultural emissions is calculated as:

$$L_{cum,agr} = \sum_{t=T-RT}^{T} (l_{river,t,agr} + l_{air,t,agr})$$

where:

$L_{cum,agr}$	cumulative agricultural nutrient load to the Baltic Sea (expressed in t N or t P)
$l_{air,t,agr}$	deposition of airborne agricultural emissions on the Baltic Sea (expressed in t N/y or t P/y)
$l_{river,t,agr}$	riverine agricultural nutrient load to the Baltic Sea (expressed in t N/y or t P/y),
RT	nutrient residence time in the Baltic Sea
T	most recent year, for which data on nutrient load is available

Residence times of N and P are estimated with 1-9 years for N and 35-49 for P based on studies by Savchuk (2018) and Radtke et al. (2012). It should be noted that assuming constant residence times over time constitutes a model simplification. In reality, nutrient residence times are dependent on trends in inflows of marine water and weather conditions (e.g., temperature and wind) HELCOM (2018b). Not least, residence times are affected by nutrient levels themselves, for instance through biological feedbacks.

HELCOM provides a time series of riverine and direct nutrient inputs, atmospheric N deposition as well as water flows to the Baltic Sea and atmospheric N emissions in the catchment since 1995 (Svendsen and Gustafsson 2019, Gauss and Bartnicki 2018, Gauss et al. 2018). Measurements of atmospheric deposition of P are very limited, which is why HELCOM assumes a constant deposition rate of 5 kg P/km^2 (HELCOM 2015). Prior to 1995, a time series of riverine P inputs and water flows based on Savchuk et al. (2012) is available in McCrackin et al. (2018). In addition, HELCOM conducts Pollution Load Compilations (PLC) at regular intervals, in which the state of nutrient loads and their sources are analysed in more detail. These are used to estimate the share of the cumulative nutrient load that can be attributed to agriculture. However, the analysis is restricted to HELCOM contracting parties; thus, transboundary agricultural emissions from non-riparian countries within the Baltic Sea catchment (primarily Belarus) have not been considered. Table 12 provides an overview of the available data on riverine agricultural emissions. For years without specific information, values are determined via linear inter- or extrapolation. Airborne agricultural N emissions deposited in the Baltic Sea are calculated using a general factor of 64-71% of reduced N deposition based on HELCOM (2015). Due to lack of data HELCOM (2015) treats P deposition as natural background input so that atmospheric P deposition of agriculture emissions is not considered in the current assessment. Similarly, agricultural emissions deposited on river surfaces and subsequently transported to the Baltic Sea are neglected.

Table 12: Reported riverine agricultural nutrient loads, absolute (Mt N, Mt P) and relative to total riverine load (%N, %P).

Year	Mt N	%N	Mt P	%P	Reference
1985			0.019		HELCOM 2003
1995			0.017		HELCOM 2003
2000			0.016		HELCOM 2003
2006		36–62%		34–55%	HELCOM 2015
2014	0.18–0.23		0.005–0.006		HELCOM 2018g

However, like estimates of nutrient residence times, all of these input data are subject to large uncertainties. Especially for the early years of the time series missing data had to be replaced with estimates. Moreover, nutrient loads are partly derived from measurements of discharges and nutrient concentrations at river mouths, partly from modelling and both measurement frequency and modelling approaches differ between different stations/countries and have changed over time (Gustafsson 2020). Therefore, different combinations of input data and calculation paths are used to compile a likely range of cumulative agricultural nutrient loads. More details on the calculation method are provided in Tanzer et al. (2021).

Nutrients present on land

Due to past overfertilisation, large amounts of P have accumulated in arable land and grassland in the Baltic Sea region, part of which is strongly bound in soil (stable pool), whereas part is prone to leaching and thus will contribute to the marine P load in the future (mobile pool). Based on studies by McCrackin et al. (2018) and Bouwman et al. (2013) the nutrient load that has accumulated in the mobile pool can be estimated with 17-19 Mt P. For the period 1900–2013 McCrackin et al. (2018) determined annual leaching rates of 0.08% and 3.17% from the mobile pool to the Baltic Sea and from the mobile to the stable pool, respectively. Under the simplified assumption that these rates are stable over time, P leaching to the Baltic Sea until the mobile pool is depleted to less than 1 t of P can be modelled as:

$$PM_{t+1} = PM_t * (1 - r_{BS} - r_{SP})$$

where:

PM_t mobile pool in year t

r_{BS} leaching rate from the mobile pool to the Baltic Sea (0.08% according to McCrackin et al. 2018)

r_{SP} leaching rate from the mobile pool to the stable pool (3.17% according to McCrackin et al. 2018)

It should be noted that P leaching from agricultural soils to the Baltic Sea is correlated to riverine discharge. As future discharge levels remain unknown, discharge is assumed to equal the long-term average 1900–2013 as reported in McCrackin et al. (2018). Changes in average discharge, e.g., due to climate change, are therefore neglected in the present study. More details on the calculation method are provided in Tanzer et al. (2021).

Total agricultural legacy nutrient loads

Based on the calculations described above the agricultural legacy nutrient loads in the Baltic Sea as of 2017 amount to 0.5–4.0 Mt N and 0.3–1.2 Mt P. Hence, agriculture is responsible for 30–40% of the cumulative load (see Table 13). Figure 14 shows the distribution of nutrient loads between the different subbasins of the Baltic Sea based on Savchuk (2018). In addition, of the 17–19 Mt P that have accumulated in mobile pools on agricultural land within the catchment, 0.4–0.5 Mt would leach to the Baltic Sea over a period of approximately 500 years. 96% of this load occur in the initial 100 years, as can be seen in Figure 15.

Table 13: Cumulative nutrient loads over nutrient lifetime in the Baltic Sea and parts of the load that can be attributed to agricultural emissions of HELCOM contracting parties based on historical emission data.

	Mt N	**Mt P**
Nutrients in the sea		
total cumulative load	**2.003–9.792**	**0.868–3.254**
cumulative agricultural load	**0.541–3.958**	**0.262–1.162**
of this: riverine	0.128–3.100	0.262–1.162
of this: deposition	0.413–0.858	0
Nutrients on land		
cumulative agricultural load	**neglected**	**0.418–0.468**

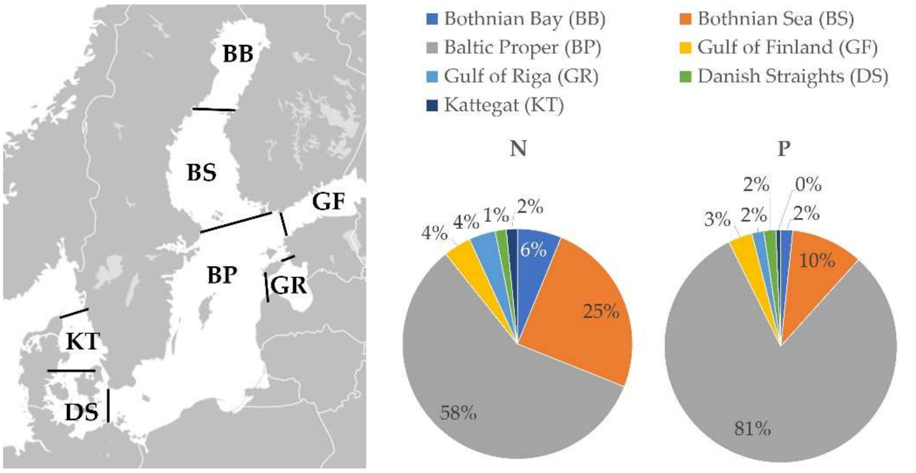

Figure 14: Overview of Baltic Sea subbasins (left, background map from Maix (2007)) and distribution of nutrient loads based on Savchuk (2018) (right).

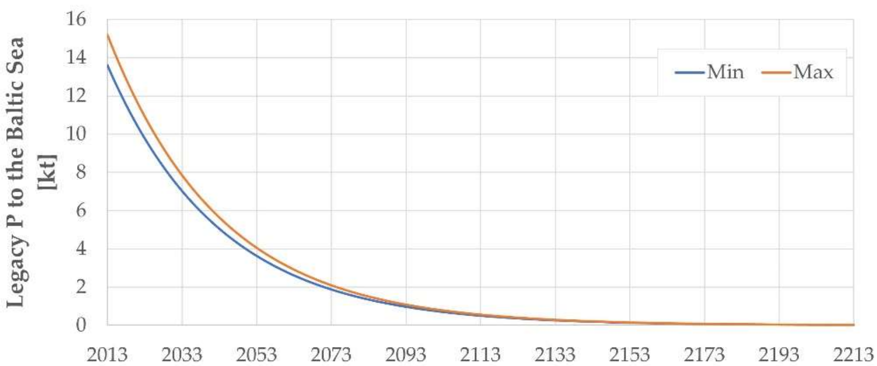

Figure 15: Leaching of legacy P from the mobile P pool in agricultural soils to the Baltic Sea during the initial 200 years under to different assumptions of initial magnitude of the mobile pool (PM min and PM max). Calculation based on McCrackin et al. (2018) and documented in Tanzer et al. (2021).

While the estimate for the total N load corresponds well with the model by Savchuk (2018) (6 Mt), the total P load in the present study is considerably higher (0.8–3.1 Mt vs. 0.7 Mt), taking into account that the lower number refers to a reference time of 35 years instead of 49 as applied by Savchuk (2018). In general, the wide ranges of the present estimates reveal that even in a comparatively well studied and monitored region like the Baltic Sea catchment, nutrient loads are associated with high uncertainties. As shown in Tanzer et al. (2021) uncertainties of the total nutrient emissions provided by HELCOM (Sevendsen and Gustafsson 2019, Gauss and Bartnicki 2018, Gauss et al. 2018) dominate over uncertainties related to nutrient residence times and methodological uncertainties of determining the agricultural share in total loads. Thus, there is a need to further harmonise measurement and modelling techniques between the different HELCOM parties and to increase understanding of emission and retention mechanisms as well as the complex ecosystem interactions governing eutrophication (HELCOM 2018c).

Furthermore, it should be minded that transboundary loads from non-HELCOM contracting parties are not included in the riverine agricultural loads due to lack of data (legacy P stored in agricultural soils also includes Belarus). Moreover, like current HELCOM assessments, the present study assumes steady state conditions in the Baltic environment (see Chapter 4.1.1). It is likely that nutrient residence times both in sea and on land as well as annual "natural" background loads change in the future and efforts to combat eutrophication will have to be reinforced compared to current projections.

4.1.4. Monetising agricultural legacy nutrient loads

Recently, measures tackling the legacy nutrient load in the Baltic Sea have attained increasing attention, including attempts to extract nutrients from the sea via mussel farming (Baltic Ecomussels 2013, NutriTrade 2018a, Schultz-Zehden et al. 2019, Petersen et al. 2020), targeted fishing of cyprinids (Mäki 2018) and harvesting of (naturally occurring or cultivated) algae (Schultz-Zehden and Matczak 2012, Olsson et al. 2013), as well as efforts to prevent P release from bottom sediments via dredging (Simonsson 2014), deep water oxygenation (Rantajärvi 2012, Stigebrandt 2014, Stigebrandt et al. 2015) and injection of aluminium (Rydin and Kumblad 2014, Kumblad and Rydin 2018) or marl (Blomqvist 2014). Meanwhile, legacy P on agricultural land is mainly addressed by structural liming (Kumblad and Rydin 2018, Pakalniete and Krumina 2020) or gypsum amendment (Ollikainen et al. 2018, NutriTrade 2018b), both aiming at binding P in more stable forms in the soil. Costs and potential impacts in terms of nutrient abatement of these measures have been estimated in various scientific articles and project reports. Therefore, it was decided to estimate costs of agricultural legacy nutrient loads as abatement cost based on this information. As abatement costs studies on emission reduction (e.g. BalticSTERN Secretariat 2013, Ahlvik 2014, Hasler et al. 2014) attribute the full costs alternately to N and P although most measures have effects on both nutrients and because not all measures tackling legacy nutrient loads are aimed at the N load, costs in this chapter are expressed as EUR per kg P removed or immobilised. This ensures comparability across measures and between studies.

However, a literature review of 45 studies (see Tanzer et al. 2021) revealed that most studies refer to pilot experiments or small-scale implementations with site-specific costs and impacts that cannot be transferred to the whole region. An upscaling to larger areas with abatement potentials above 100 t P has only been undertaken in eight studies, as shown in Table 14.

Table 14: Costs and potential impacts (Mt P removed or immobilised) of soil and sea-based measures

Measure	Extent	Potential Impact [Mt P/a]	Costs [€/kg P]	Repetition	Reference
Soil-based					
Gypsum amendment	Finland (potential extension to SE, DK, DE and PL)	0.0002–0.0003 (0.001–0.002)	55–86	5 years	Ollikainen et al. 2018, NutriTrade 2018b
Structural liming[1]	Arable land with clay content >20% in Swedish North and South Baltic Sea Water Districts	0.0001	222	10–30 years	Kumblad and Rydin 2018
Sea-based					
Aluminum treatment[1]	Swedish Coastal area of the Baltic Proper	0.0005	89	one-time	Kumblad and Rydin 2018
Deep water oxygenation[2]	Baltic Proper	0.060–0.092	2–4	one-time	Eriksson and Kullander 2013, Stigebrandt and Andersson 2020
Deep water oxygenation[3]	Bornholm Basin (Baltic Proper)	0.005–0.008	2–5	one-time	Stigebrandt 2014, Eriksson and Kullander 2013
Deep water oxygenation[4]	Finnish parts of the Gulf of Finland	0.00007–0.00012	28–48	annual	Rantajärvi 2012
Deep water oxygenation	large-scale, not specified	not stated	2–75	not stated	Vahanen Environment OY 2018
Blue mussel farming	Baltic Proper	0.010	not stated	1–2 years	Kotta et al. 2020
Blue mussel farming	Bothnian Sea, Bothnia Bay, Gulf of Finland, Gulf of Riga	0.001	not stated	1–2 years	Kotta et al. 2020

[1] Costs are given in SEK in Kumblad and Rydin (2018). The annual exchange rate for 2011 (1 SEK = 0.1108 €) is used for conversion.
[2] Costs are given in SEK in Eriksson and Kullander (2013). The annual exchange rate for 2013 (1 SEK = 0.1156 €) is used for conversion. Extent according to Eriksson and Kullander (2013) (0.092 Mt P) dates back to 2005; while a more recent study by the same research group estimates 0.060 Mt P (Stigebrandt and Andersson 2020). No repetition is considered, as oxygenation is assumed to only be necessary for 10–15 years Baltic Sea Restoration (2017). The depreciation time is 20 years (Eriksson and Kullander 2013).
[3] No repetition is considered, as oxygenation is assumed to only be necessary for 10–15 years Baltic Sea Restoration (2017). The depreciation time is 20 years (Eriksson and Kullander 2013).
[4] Rantajärvi (2012) assume that oxygenation has to be conducted permanently to prevent remobilisation of P from the sediments. The depreciation time is 20 years. A pessimistic scenario, where no reduction in the internal P source can be achieved is not considered, as no unit costs can be calculated in this case.

Kotta et al. (2020) do not provide cost estimates for the regional upscaling of mussel farming. However, cost estimates for numerous individual farms in different region have been reported as shown in Table 15. Contrary to soil- and sediment-based measures, mussel farming is not only a remediation measure, but could also create economic benefits and is already commercially undertaken in the western Baltic Sea (Ozolina and Kokaine 2019). Although mussels from the central and inner Baltic are probably not suitable for human consumption due to their decreasing size following the salinity gradient, there are numerous alternative market opportunities including processing to feed or fertiliser and production of environmentally friendly anti-corrosive products, adhesives, or human nutraceuticals (Schultz-Zehden and Matczak 2012, Schultz-Zehden et al. 2019). Schultz-Zehden et al. (2019) estimate that such markets could become economically viable if production costs do not exceed 0.1 €/kg (translating to 114 €/kg P in the outer-, 189 €/kg P in the central-, and 250 €/kg P in the inner Baltic (Kotta et al. 2020)). Considering the production costs in Table 15, it is thus evident, that mussel farms will at least partly depend on external subsidies. Required subsidies (i.e. production costs minus production costs under economic viability are also shown in Table 15.

Table 15: Production costs and required subsidies of blue mussel farming in different regions of the Baltic Sea (Tanzer et al. 2021). Lower value of production costs in the Outer Baltic refers to production costs under economically profitable production.

Region	EUR/kg P (Production Costs)	EUR/kg P (Required Subsidies)
Outer Baltic (Kattegat and Belt Sea)	114–2846	0–2732
Central Baltic (Baltic Proper)	250–5230	69–5041
Inner Baltic (Bothnian Bay, Bothnian Sea, Archipelago Sea, Gulf of Finland, Gulf of Riga)	728–21 300	131–21 050

Based on the information provided in Table 14 and Table 15, an abatement scenario for agricultural legacy nutrient loads as shown in Table 16 was developed.

Table 16: Abatement costs of agricultural legacy loads. N abatement by mussel farming is calculated from the P:N-ratio in harvested mussels (Kotta et al. 2020) and therefore exceeds the agricultural legacy load shown in Table 13.

Measure	P Abatement [Mt]	N Abatement [Mt]	Costs [Billion €]
Deep water oxygenation	0.060–0.092	0	0.184–4.500
Soil gypsum amendment	0.145–0.183	0	0.010–0.013
Mussel farming	0.406–1.425	5.275–18.511	32.422–111.455
Sea load	*0.170–1.102*	*2.210–14.315*	*14.055–86.311*
Soil load	*0.236–0.323*	*3.065–4.196*	*18.367–25.144*
Total	**0.680–1.630**	**5.275–18.511**	**32.616–115.967**

As the most cost-effective measures in Table 14, it is assumed that deep water oxygenation and soil gypsum amendment are assumed to be implemented to their full extent[4]. The remaining abatement of legacy nutrients in the sea (i.e. remaining sea based load and leaching from mobile soil pools that cannot be prevented by soil gypsum amendment) is assumed to be reached via blue mussel farming. Abatement potentials estimated by Kotta et al. (2020) do not refer to a scenario where all area considered suitable for mussel farming is used, but to an extent to which mussel farms could easily be implemented under the current spatial planning regime of the Baltic Sea. Furthermore, Kotta et al. (2020) only study potentials of mussel farming in the central and inner Baltic, so that for the outer Baltic current production levels were assumed to remain stable. Hence, to remove the remaining agricultural legacy loads mussel farming would have to be conducted for 56–166 years in the inner-, 33–120 years in the central- and 144–428 years in the outer Baltic. Because of the conservative estimation of mussel farm extent, it is assumed that only locations with the most favourable conditions and thus lowest costs are used. Costs per kg P removed are therefore set to the lower end of the ranges for required subsidies shown in Table 15. Overall abatement costs amount to EUR 33–116 billion (see Table 16). Details on cost calculations can be found in Tanzer et al. (2021).

It should be born in mind that abatement costs presented here merely constitute a first rough estimate to monetise the impact of agricultural legacy nutrient loads and more research is needed to develop a more realistic scenario. In particular, values given in Table 16 should not be used for the development of concrete abatement strategies. For instance, to date the Baltic Proper is the only subbasin for which potentials of deep water oxygenation have been estimated. However, as shown by Rantajärvi (2012)

[4] In fact, the costs for structural liming and gypsum amendment seem to be in a similar range, considering the different lifetimes of the measures. Soil gypsum amendment has been chosen for further analysis as abatement potentials have been estimated for a larger region.

deep water oxygenation may also be possible in other subbasins and the potentials of this measure in the present scenario are most likely underestimated. Similarly, mussel farming in the central and inner Baltic is still in the pilot stage and has therefore generally not been optimised for nutrient extraction. With increasing experience, technological advances and once a critical size allowing for industrial production and economies of scale is reached, higher yields and lower production costs may be possible (NutriTrade 2018a, Schultz-Zehden et al. 2019, Ozolina and Kokaine 2019). Furthermore, some potentially viable measures such as targeted fishing or harvesting of algae have completely been neglected in the present analysis because they have not been demonstrated beyond pilot experiments in single locations (Mäki 2018, Schlultz-Zehden and Matczak 2012, Olsson et al. 2013).

On the other hand, as nutrient removal progresses, ecological conditions in the Baltic Sea might change, resulting in lower food availability and thus lower abatement potentials of mussel farming in the future. This may necessitate a relocation of mussel farms and entail higher production costs of mussel farming over time. Especially if spatial conflicts with recreation, aquaculture, environmental protection, transport or energy generation forces mussel farms to move further away from shorelines, production costs, at least with current technology, will increase considerably (SUBMARINER Network Mussels Working Group 2019).

Moreover, long-term effects and potential risks of large-scale implementations of the measures listed in Table 14 are not sufficiently studied to date. Oxygenation pumping for instance may destroy the thermal and salinity stratification of the ocean and cause the release of contaminants previously bound in the sediments (Conley 2012, Vahanen Environment OY 2018). Similarly, intensive mussel farming could cause unpredictable and severe changes in marine biodiversity (Kotta et al. 2020), although the risk of oxygen deficits following sedimentation of organic material underneath the farms or competition with fish populations is gauged marginal in recent studies (SUBMARINER Network Mussels Working Group 2019). Sites for soil gypsum amendment also have to be carefully selected to prevent contamination of lakes, groundwater or ecologically valuable sites with sulphate (Ollikainen et al. 2018).

Finally, the abatement scenario assumes stable socio-economic and environmental conditions in the Baltic Sea region, which especially considering the projected duration of mussel farming of up to 500 years, constitutes a severe limitation of the present study. Nevertheless, as it is difficult to determine which of the effects described above will prevail in the long run, estimations based on current conditions are the best available ones to date.

4.1.5. Case study conclusions

Table 17 provides an overview of costs of eutrophication in the Baltic Sea attributable to fertilising practices as presented in Chapter 4.1.4 and derived from past cost estimations. In addition, a first tentative assessment linking pressures to and dependency on an intact marine environment for different economic sectors in Sweden has recently been undertaken in the HELCOM SPICE project (Ahtiainen et al. 2017), albeit only in a qualitative form. In this analysis, tourism seems to be the most crucial sector to address as it both generates high economic value and is highly dependent on ecosystem services, but at the same time causes considerable damage to the marine environment.

Table 17: Cost estimates of eutrophication in the Baltic Sea attributable to fertilising practices. Based on results in Chapter 4.1.3. it is assumed that agriculture accounts for roughly one third of present and past nutrient loads. Total costs including emissions from all sources as provided in the different studies are shown in brackets. Regarding abatement cost studies the term "mitigation" refers to studies quantifying costs of reducing present nutrient loads, whereas "remediation" refers to end-of-pipe removal of nutrients from past emissions from the sea. BSAP: Baltic Sea Action Plan. BAU: Business-As-Usual.

Reference	Method	Impact	Costs [billion EUR/year]	Costs [EUR/kg P]
Gren 2008	Abatement costs (mitigation)	Achievement of BSAP 2007 goals (reduction of 21% N load and 48% P load compared to 2005, i.e. 0.17 Mt N/year and 0.02 Mt P/year)	0.54 (1.62)	81
Gren 2008	Abatement costs (mitigation)	Reduction of P load by 70% compared to 2005, i.e. 0.03 Mt P/year	1.33 (4)	147
The Swedish Environmental Protection Agency 2009	Economic losses (survey among tourism representatives)	More widespread occurrence of algal blooms	0.16-0.24 (0.48-0.72)[2]	Not stated
Hasler et al. 2012[1]	Abatement costs (mitigation)	Reduction of nutrient loads by 0.21 Mt N/year and of 0.01 Mt P/year (maximum achievable BSAP 2007 goals for the studied measures)	1.56 (4.69)	375
BalticSTERN Secretariat 2013	Abatement costs (mitigation)	Reduction of nutrient loads by 0.11 Mt N/year and of 0.01 Mt P/year (maximum achievable BSAP 2007 goals for the studied measures)	0.78 (2.34)	205
Ahtiainen et al. 2014	Stated preference (WTP survey)	Achievement of BSAP 2007 goals compared to BAU (comparison of 2050 situation)	1.25-1.46 (3.76-4.38)	Not stated
Ahlvik et al. 2014	Abatement costs (mitigation)	Reaching of good eutrophication status by 2050	0.50 (1.49)	Unknown
Hasler et al. 2014[1]	Abatement costs (mitigation)	Reduction of nutrient loads by 0.21 Mt N/year and of 0.01 Mt P/year (maximum achievable BSAP 2013 goals for the studied measures)	1.39 (4.17)	333
Wulff et al. 2014[1]	Abatement costs (mitigation)	Reduction of nutrient loads by 0.13 Mt N/year and of 0.01 Mt P/year (maximum achievable BSAP 2007 goals for the studied measures)	1.55 (4.65)	386
Czajkowski et al. 2015	Revealed preference (Travel cost survey)	Undefined improvement of water quality	0.34-0.72 (1.02-2.16)	Not stated
Dahlgren et al. 2015	Economic modelling	Achievement of BSAP goals by 2025 compared to a "shipwreck state" with further increasing nutrient loads (comparison of 2015-2030 situation)[4]	9.00 (27.00)	Not stated
Chapter 4.2. and 4.3. Tanzer et al. (2021)	Abatement costs (remediation)	Removing legacy nutrient loads (5.28–18.51 Mt N and 0.68-1.63 Mt P)	0.07-0.21[3] (0.21-0.63)	45-67

[1] Studies based on the same model (BALTCOST)

[2] According to the survey 10-15% losses in sales are expected. Annual costs are calculated from the annual GVA in the coastal tourism accommodation sector in 2015 (HELCOM 2018a).

[3] Annual costs correspond to the average over the whole remediation period (ca. 500 years). Average costs during the initial 15 years of measure implementation amount to EUR 0.79-1.11 billion per year and decrease subsequently.

[4] Considering the long residence time of P in the Baltic Sea (see Chapter 2.2.) manifestation of benefits of nutrient reductions by 2030 might not be realistic.

To put costs in Table 17 into perspective, Table 18 lists the GVA and number of people employed in different economic sector using the Baltic Sea.

Table 18: Gross value added (GVA) / Added factor costs and number of persons employed in different economic sectors dependent on the Baltic Sea in the coastal states except for Russia (HELCOM 2018a). Note that this table is only indicative and not exhaustive, due to different reference years (2014/2015), and lack or confidentiality of data for some sectors and/or countries. For details see HELCOM (2018a).

Activity	Annual GVA [billion EUR]	Number of persons employed
Fishing	0.12	9 040
Marine aquaculture	0.01	256
Coastal tourism – accommodation	4.80	176 321
Shipping – freight transport	4.44	22 278
Shipping – passenger transport	1.59	24 572

As also explained in Chapter 2.2, due to different perspectives (e.g. analysis of economic or intrinsic values), methods (e.g. modelling or survey), scenarios analysed (e.g. different levels of N and P load reduction), and monetary unit (e.g. different reference years, adjustment to purchasing power) in different studies, they can neither be directly compared, nor combined. In particular, Table 17 should not lead to the conclusion that end-of-pipe remediation of nutrient loads by sea-based measures are more cost-efficient than prevention of nutrient emissions on land. It has to be born in mind that even although only covering part of the total legacy nutrient load in the sea remediation activities are projected to last for half a century and it yet has to be assured whether implementation will be feasible without causing spatial conflicts or disturbances to the ecosystem. Especially in light of the unknown consequences of climate change on nutrient residence times it would be negligent to rely only on end-of-pipe measures and not tackle nutrient emissions at the same time. On the other hand, land-based measures will not show immediate effects if legacy nutrient stocks in the sea are not addressed.

It is also evident that even in a comparatively well studied and monitored region like the Baltic Sea catchment past and present nutrient loads can only be reconstructed with high uncertainties and monetising their impacts is even more difficult. Nevertheless, from Table 17 it is clear that past and present fertilising practices in the catchment put substantial pressures on the Baltic Sea, which are in a similar range of both GVA generated by using the sea (Table 18) and fertiliser contribution to agricultural GVA (EUR 0.67 billion). The latter is based on an annual mean agricultural GVA of EUR 30.99 billion in the region (Tanzer et al. 2021), the mean share of fertilisers in intermediate agricultural consumption in the HELCOM EU states and the assumption that 75% of agricultural GVA stem from human input (see Chapter 3.2).

Finally, the present case study also shows that while mitigating eutrophication impacts entails substantial costs, if implemented wisely measures could also open new business opportunities and create sources of economic income.

4.2. Case Study: Trade-off of health costs of the ammonia (NH_3) emissions in agriculture in fertiliser choices in Flanders

4.2.1. Review of literature on health costs on ammonia emissions

Increasing nitrogen inputs from fertilizer applications have contributed to higher agricultural outputs in Europe in recent years. They have also resulted in the release of nitrogen into the environment, specifically through ammonia (NH_3) emissions into the atmosphere. Ammonia (NH_3) is a colorless gas and consists of one nitrogen and three hydrogen atoms. It dissolves easily in water to form ammonium hydroxide. It is also used as a refrigerant gas, to purify water supplies, and in the manufacture of plastics, pesticides, and other chemicals. Over 81% of global NH_3 emissions are attributed to agriculture (Van Damme et al. 2021). The main sources of NH_3 emissions from agriculture are livestock and animal production, manure handling and storage, livestock housing, and the application of manure/slurry and synthetic fertilizers to land (Figure 16). This note provides a synthesis of the existing evidence on the cost of ammonia emissions from agriculture.

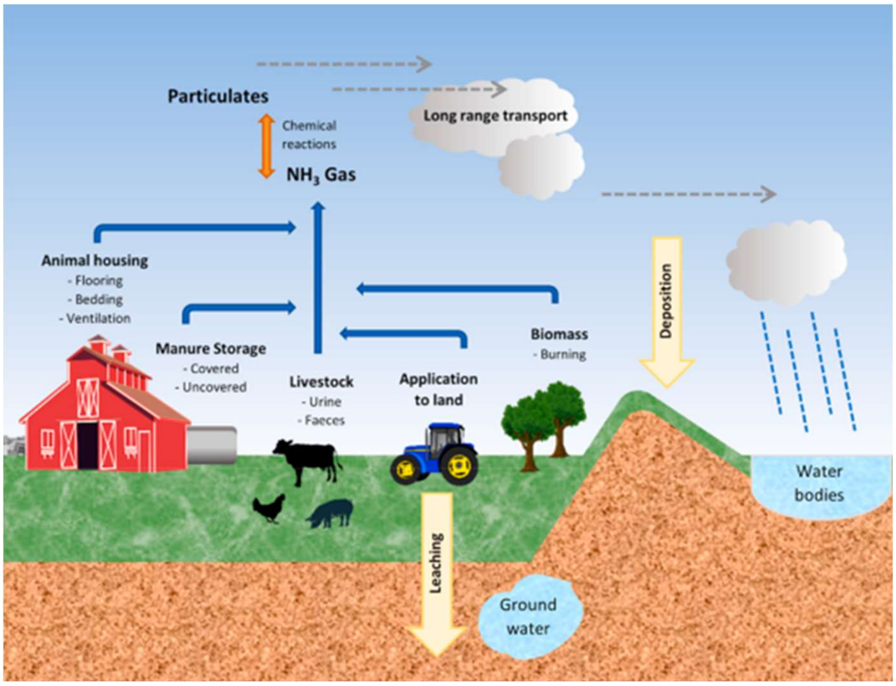

Figure 16: Potential sources of NH3 from agriculture (Source: Wyer et al. 2022)

Costs of ammonia emissions from agricultural on health and biodiversity

One of the bigger costs of NH_3 emissions is its impact on health. Indeed, NH3 is considered as an irritant. When exposed to excessive amounts of NH_3, it can have serious consequences for human health (Wyer et al. 2022). Exposure to extremely high levels of NH_3 in the agriculture sector is uncommon, occurring most often during farming accidents. However, exposure to lower concentrations over longer periods of time, on the other hand, may have a negative impact on human health (Lelieveld et al. 2015). The most commonly known health impacts are eye, nose, and throat irritation, headache, nausea, diarrhea, hoarseness, sore throat, cough, chest tightness, nasal congestion, palpitations, shortness of breath, stress, drowsiness, and mood changes (Domingo et al. 2021; Brunekreef et al. 2015; Lelieveld et al. 2015; Balasubramanian et al. 2021).

Although direct exposure to agricultural NH_3 emissions can be problematic for human health when assessed as a precursor to fine particulate matter (PM2.5), which has a much greater potential impact on human health. World Health Organization (2021) describes PM2.5 as particles that are small enough to penetrate the thoracic region of the respiratory system once inhaled. Its exposure has also been linked to chronic obstructive pulmonary disease (COPD), lung cancer, and premature death (Lelieveld et al. 2015; Yu et al. 2000). Numerous precursors to PM2.5, such as nitrogen oxide (NO_x) and sulphur dioxide (SO_2), are well-regulated; however, studies have shown that NH_3 is the precursor that has the greatest impact on the formation of this pollutant (Thakrar et al. 2020). Consequently, it is essential to comprehend how agricultural NH_3 is emitted into the atmosphere. According to Van Grinsven et al. (2013), the social cost of the impacts of agricultural NH_3 emissions in the European Union in 2008 was estimated between 10 and 120 billion euros per year, of which 5–65 billion euros were associated with air pollution effects on human health. As stated by McCubbin et al. (2002) and Muller et Mendelsohn (2010), the average annual health cost of 1 kilogram of NH_3 emitted into the atmosphere in the United States ranges from $0.01 to $73 in 2006, depending on the valuation method, which is 2 and 9 times higher than the cost of 1 kg of SO_2 and NO_x, respectively. On the other hand, Paulot et Jacob (2014) estimates that the resulting annual health cost of PM2.5 is 100 US$ (2006) per kg of NH_3. This variation is partially attributable to the spatial distributions of various NH_3 sources, with sources located closer to population centers having a greater impact. The cost can also vary based on the type of source.

Ammonia emissions have also significant implications on biodiversity. An important impact of ammonia pollution on biodiversity is the impact of nitrogen accumulation on the diversity and composition of plant species in habitats that are impacted. In a nitrogen-rich environment, species adapted to high nutrient availability thrive and outcompete more sensitive, smaller, or rarer species. Ammonia pollution also influences species composition via soil acidification, direct toxic damage to leaves and plants. Therefore, herbivorous animals are susceptible to the effects of ammonia pollution (Borer et al. 2014). Ammonia also affects freshwater ecosystems through direct agricultural runoff leading to eutrophication (accumulation of nutrients leading to algal growth and oxygen depletion) and also has toxic effects on aquatic animals with porous skin. According to Guthrie et al. (2018), the impact of biodiversity loss due to ammonia emissions on the United Kingdom could be valued between £0.20 and £4 per kilogram of ammonia. Combining this with the quantified health impacts, their results estimate that the total cost of ammonia's health and biodiversity impacts in the United Kingdom is £2.50 per kilogram of ammonia (with a range of £2 to £56 per kilogram).

4.2.2. Simulation of trade-off health costs – economic profit: case study of Flanders

The current situation of Flanders is one with the production of manure from livestock production exceeding the crop requirements for P. Almost no mineral P fertilizers are used and there is a surplus of P fertilizer based on the manure production. In 2020 in Flanders, a total of 47.3 million kg P_2O_5 could have been applied as fertilizers while there was 60.5 million kg P_2O_5 present in the livestock manure. The region fertilization policies enforce P fertilization standards, which means that the surplus needs to be processed and exported. The most common option is to export poultry manure which does not yet fully sets the P balance straight. The following option on the economic cost ladder is to apply solid – liquid separation to pig slurry, which is also very commonly applied and whereby the solid fraction is exported after drying of composting.

The remaining liquid fraction could be applied on land as

- an N fertilizer,
- could undergo stripping to obtain a recovered mineral or renure fertilizer or
- could undergo nitrification denitrification to remove also some of the surplus N.

These three options form the core of our analysis in this case study.

There are varying economic costs and ammonia emission related to these three options. The direct application of the liquid fraction of pig slurry is cheap but the composition is less certain than synthetic or recovered fertilizer in the form of ammonium nitrate of sulfate. In addition, there is a higher risk of ammonia volatilization during storage and application of the liquid fraction of pig slurry.
The option of stripping a part of the N in the form of ammonium sulfate reduces the risk of ammonia losses and increases the certainty of the composition of the fertilizer for the farmer. However, there is a cost of the stripping process associated with it.

The third option minimizes the risk of ammonia losses by nitrification-denitrification of the liquid fraction and the application of calcium ammonium nitrate as fertilizer. The least circular approach has also the highest cost associated with it because of the biological treatment of the liquid fraction and the cost of the synthetic fertilizer.

The trade-off between these three options depends also on the yield response to fertilization, the weather variability, the economic value of the crop and all the costs of the above mentioned parameters and ammonia volatilization coefficients for the different unit processes.

The following scheme (Figure 17) shows the interactions of these trade-offs between economic costs and ammonia emission and how they could be simulated.

To illustrate these trade-offs, the scheme below is implemented in a stochastic mathematical programming model that either optimizes pure economic profit or also taking the external costs of ammonia emissions into account. The stochasticity is driven by weather (20 year with daily variation), sensitivity to economic prices and the health impact.

Given the large variation in estimated health costs, the simulation will also be performed with the same large variation as reported in literature.

Each of the arrows in the scheme shows a choice variable in the model that is fed with discrete point simulation of a crop yield model.

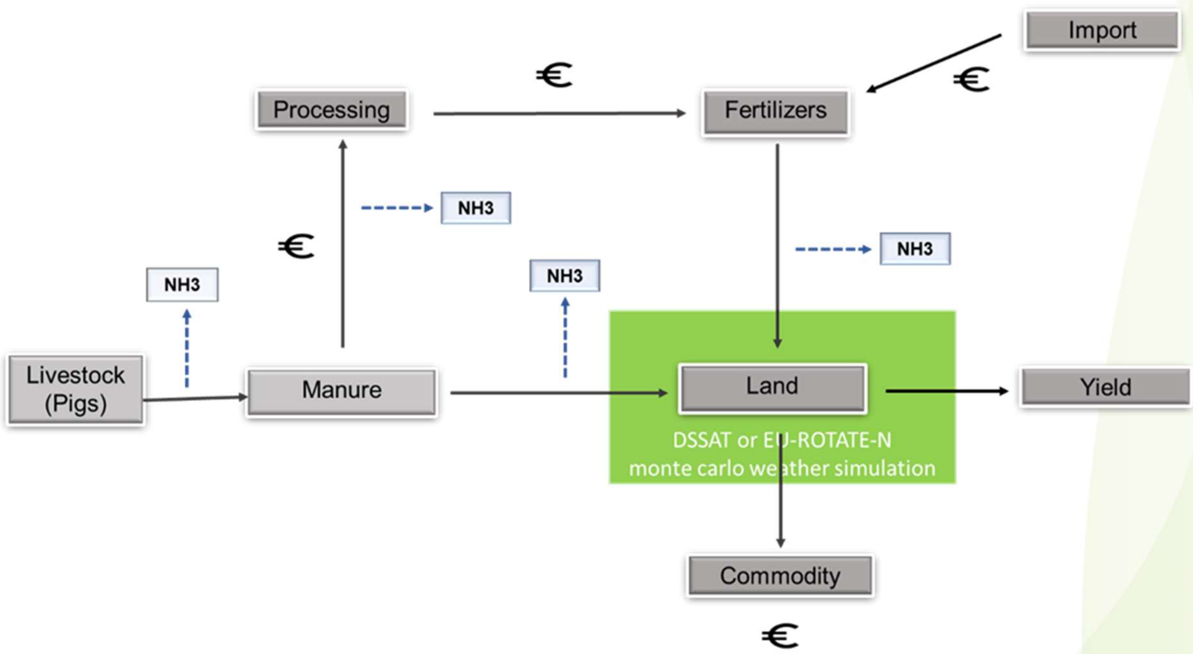

Figure 17: Scheme of the stochastic mathematical programming model.

The simulation focuses on the fertilization decision of 1 ha of maize in Flanders and the related production and processing cost of the fertilisers and the manure. We assume that the liquid fraction of pig slurry is available at a rate of 170 kg N/ha and that this liquid fraction needs to be processed if it is not used as a fertilizer.

The simulation uses a discrete crop yield function of maize given crop and fertilizer variability on a sandy soil. The assumed processing cost is 2.5 euro /kg N processed in nitrification denitrification, the costs of stripping is 3.5 euro/ kg N and the cost of synthetic fertilizer is 3 euro/kg N. Transport costs and field application costs are not considered because they are assumed to show only minor differences between the varying field applications. Stripping is limited by a technical limit of 59 kg of the total of 170 kg of N available. The field application of the liquid fraction of pig slurry is assumed to result in 10% on N basis ammonia volatilization. This can be considered as an efficient field application because the literature report volatilization losses of slurry spreading of up to 50%. These losses depend on the timing, the weather, technique, soil cover and the technique for field incorporation.

Table 19 shows the results of the baseline simulations with the assumed costs and benefits as stated while varying in each simulation the external costs of ammonia volatilization and the crop prices. The results clearly show that for the realistically assumed input parameters, the external costs of ammonia emissions drive the optimal solution in one or another direction.

The choice for fertilizer product seems to be driven more by the wide range of in the literature reported external cost of ammonia emission instead of the price of the crop. Yet, the higher the price of maize, the higher the willingness to pay for fertilizers such as synthetic fertilizers and recovered N through stripping. The choice for stripping and synthetic fertilizer is also driven by the attempt to reduce the ammonia losses from field application of liquid fraction.

Table 19: Model input parameters and results of the baseline simulations.

Model input parameters		Model results						
Ammonia external cost euro/kg Nr	**Crop output price euro/ton**	**Synthetic N kg/ha**	**Stripped N kg/ha**	**Processed N kg/ha**	**applied LF N kg/ha**	**Expected Crop yield DM ton/ha**	**Expected N residue kg/ha**	**Welfare euro/ha**
	200	30			170	10,62	29,27	2034
10	200	30			170	10,62	29,27	1864
20	200	30	8		163	10,68	28,65	1694
30	200	24	59		111	10,94	28,17	1575
40	200	24	59		111	10,94	28,17	1464
50	200	91	59	111		10,99	20,59	1440
	220	30			170	10,62	29,27	2246
10	220	30			170	10,62	29,27	2076
20	220	30	8		163	10,68	28,65	1907
30	220	24	59		111	10,94	28,17	1793
40	220	24	59		111	10,94	28,17	1683
50	220	91	59	111		10,99	20,59	1660
	240	30			170	10,62	29,27	2458
10	240	30			170	10,62	29,27	2288
20	240	30	45		125	10,93	29,16	2126

30	240	24	59		111	10,94	28,17	2012
40	240	24	59		111	10,94	28,17	1902
50	240	91	59	111		10,99	20,59	1880
	260	30			170	10,62	29,27	2671
10	260	30			170	10,62	29,27	2501
20	260	30	45		125	10,93	29,16	2344
30	260	24	59		111	10,94	28,17	2231
40	260	24	59		111	10,94	28,17	2120
50	260	91	59	111		10,99	20,59	2100
	280	30			170	10,62	29,27	2883
10	280	30			170	10,62	29,27	2713
20	280	30	45		125	10,93	29,16	2563
30	280	24	59		111	10,94	28,17	2450
40	280	24	59		111	10,94	28,17	2339
50	280	91	59	111		10,99	20,59	2319
	300	30			170	10,62	29,27	3095
10	300	30			170	10,62	29,27	2925
20	300	30	45		125	10,93	29,16	2782
30	300	24	59		111	10,94	28,17	2668
40	300	24	59		111	10,94	28,17	2558
50	300	91	59	111		10,99	20,59	2539
	320	30			170	10,62	29,27	3308
10	320	30			170	10,62	29,27	3138
20	320	30	45		125	10,93	29,16	3000
30	320	24	59		111	10,94	28,17	2887
40	320	24	59		111	10,94	28,17	2777
50	320	91	59	111		10,99	20,59	2759
	340	30			170	10,62	29,27	3520
10	340	30	8		163	10,68	28,65	3351
20	340	30	45		125	10,93	29,16	3219
30	340	24	59		111	10,94	28,17	3106
40	340	24	59		111	10,94	28,17	2995
50	340	91	59	111		10,99	20,59	2978
	360	30			170	10,62	29,27	3732
10	360	30	8		163	10,68	28,65	3564
20	360	30	45		125	10,93	29,16	3437
30	360	24	59		111	10,94	28,17	3325
40	360	24	59		111	10,94	28,17	3214
50	360	91	59	111		10,99	20,59	3198
	380	30			170	10,62	29,27	3945
10	380	30	45		125	10,93	29,16	3781
20	380	30	45		125	10,93	29,16	3656
30	380	24	59		111	10,94	28,17	3543
40	380	24	59		111	10,94	28,17	3433

50	380	91	59	111		10,99	20,59	3418

4.2.3. Case study conclusions

Within the wide range of estimated health costs in the literature, the optimal fertilization mix for a case study in Flanders differs. This means that current economic optimal choice, the application of minimally processed manure based products and synthetic fertilizers leads to a socio-economic suboptimal welfare. When internalizing the cost of ammonia emission, the optimal combination includes more synthetic fertilizer and more recovered nutrients from manure. The increasing with an increased assumed external cost of ammonia emission.

In the further steps of the Lex4bio project and nutrient policy implementation, two issues need to be considered.

First, given the strong impact of the estimated cost of ammonia pollution, more research is needed to come to a smaller range of consensus estimates of damage functions of ammonia emissions.

Second, the fact that ammonia emissions are a key driver, also from public perspective, in choice of technology for treatment, recovery and fertilization, more emphasis should be taken in the project to measure, simulate and assess the difference in ammonia emissions from the various fertilizer products, their application and their production.

4.3. Case study: Fertilisation impacts reflected in the efficiency of the Austrian Agri-environmental Programme (ÖPUL)

4.3.1. Case study description

Characteristics of Austrian agriculture

Austrian agriculture is small-structured, the average farm size was 23.6 ha in 2020 with 93% of farms family-owned and 57% managed on a sideline basis. Agriculture contributed with EUR 8.5 billion or 0.5% to the national GDP. While alpine farming dominates in the mountainous regions in the west, arable farming prevails in the east. In the transition region of the Alpine foothills, arable farming mixes with pig and poultry production. In total, agricultural land comprises 1.32 million ha arable land, 1.21 million ha pastures and 0.06 million ha permanent crops (wine and fruit trees). Especially the production of oil and protein crops have increased over the last years with Austria emerging as one of the largest soya producers in the EU. Regionality and mandatory indications of origin are important priorities in Austrian agricultural policy. With 27% Austria has the highest share of organic farmland in the EU (Federal Ministry Republic of Austria Agriculture, Forestry, Regions and Water Management 2022a, b).

In 2019 agriculture contributed 9% or 7152 kt CO_2eq to the Austrian greenhouse gas emissions. Agricultural greenhouse gas emissions have decreased by 12% since 1990, mainly due to reductions in livestock numbers and mineral fertiliser applications. Current emissions of mineral and organic fertiliser application amount to 495 kt CO_2eq and 647 kt CO_2eq, respectively. Contrary to emissions from mineral fertiliser application, which have declined by a quarter since 1990, organic fertilisers could only be reduced by 5% over the same period. Regarding emissions from manure management (974 kt CO_2eq in 2019) reductions in livestock numbers have been counterbalanced by an increase in liquid systems so that overall, no significant reduction over time can be seen (Anderl et al., 2021). In addition to greenhouse gas emissions, agriculture is also a main source of NO_x (11 kt or 9% of the

Austrian total in 2020), NMVOC (36 kt or 32% of the Austrian total in 2020), NH_3 (61 kt or 94% of the Austrian total in 2020) and PM_{10} (4 kt or 15% of the Austrian total in 2020). These emissions mainly stem from application of both mineral and organic fertilisers as well as manure management, with the exception of PM_{10} emissions, which mainly arise during tillage and harvest operations. All have experienced declines since 1990 between -8% (NH_3) and -29% (NMVOC) due to decreases in agricultural area and livestock numbers, as well as changes in feeding practices and low emission fertiliser application techniques (Anderl et al. 2022).

Annual average gross nutrient balances were around 40 kg N/ha and 1.6 kg P/ha in the period 2015-2019. A weak decline can be observed over the past 20 years. N efficiency has increased from 55% to 73% over the same period. Nitrogen surpluses are particularly high in regions with high livestock numbers, as shown in Figure 18. P balances exhibit a strong decline over the past 20 years with efficiency increasing from 61% to 92%. In high-yield years even depletion of soil P stocks may occur (Schwarzl 2021).

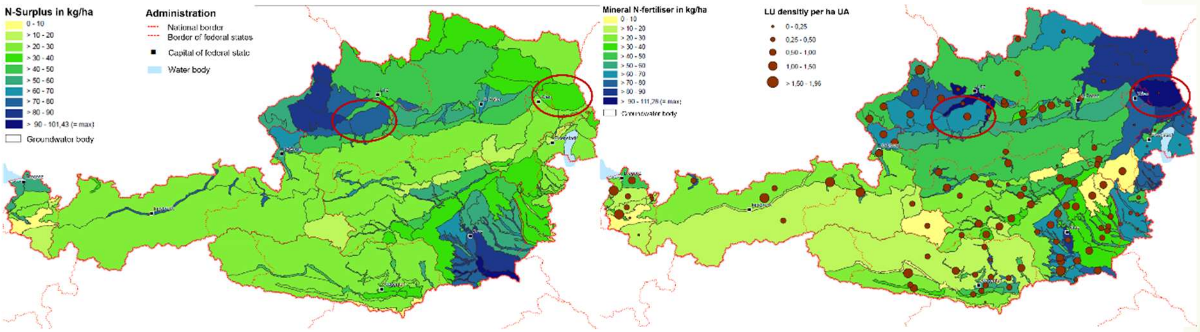

Figure 18: Average N-Surplus (left), mineral N fertiliser application and livestock unit density per utilised agricultural area in Austria for the years 2009 to 2012. Adapted from Losihandl-Weisz et al. (2013). Red circles indicate focus regions of the present case study (see Chapter 5.1.2).

The Austrian agri-environmental programme ÖPUL

The Austrian agri-environmental programme ÖPUL was established in 1995 and is currently in its fifth funding period “ÖPUL 2015”. A new funding period will start in 2023. ÖPUL 2015 aims at widespread participation across Austria for the protection of water, soil, climate, biodiversity and cultural landscape. It contains the measures in the areas of agri-environment and climate (19 measures), organic agriculture (1 measure), payments under Natura 2000 and the Water Framework Directive (2 measures) and animal welfare (2 measures). Farmers are granted premiums for each measure they implement on a per ha or per livestock unit basis. In 2021 81% of farms and 79% of agricultural areas participated in ÖPUL measures making it one of the most well accepted agri-environmental programmes in the EU. In total payments for ÖPUL measures amounted to EUR 437 million in 2021 and thus to 18% of payments in the primary sector (Federal Ministry Republic of Austria Agriculture, Forestry, Regions and Water Management 2022a). Figure 19 shows the distribution of payments, area and participating farms to the different ÖPUL measures. However, effectiveness of the payments has been questioned for at least some of the measures. For instance, under the measure “Implementation of the Water Framework Directive on agricultural land” premiums are granted for what is already prescribed in the legislation. Similarly, it has been estimated that only around a third of the increase in organic farming area can be attributed to payments under ÖPUL, while the EU organic production regulation and the resulting sales market might have played a bigger role (Niedermayr et al. 2019).

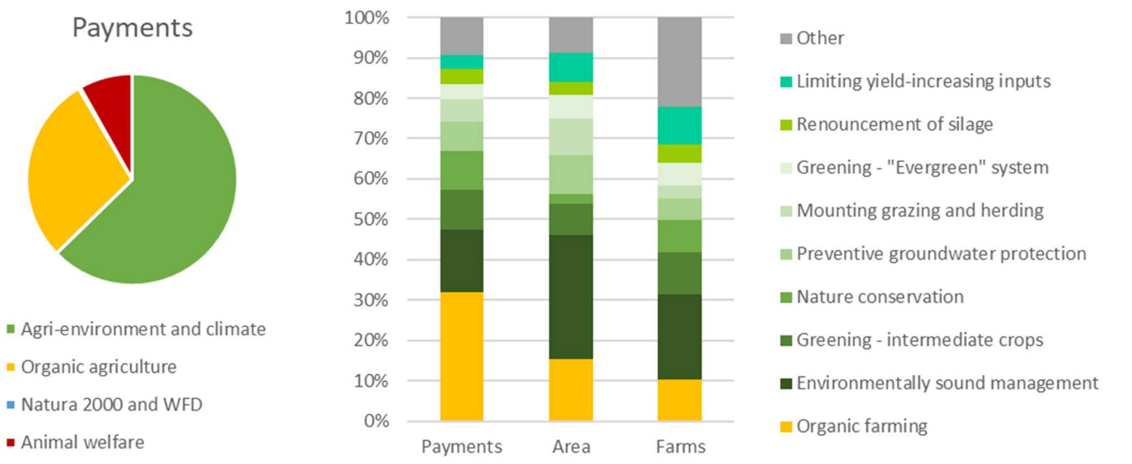

Figure 19: Distribution of payments under ÖPUL to different areas of measures (left) and share of payments, area and participating farms for area-based measures in 2021 (right). Data from Federal Ministry Republic of Austria Agriculture, Forestry, Regions and Water Management (2022a).

4.3.2. Objectives of the case study

For the Austrian case study, results from the evaluation of the ÖPUL 2015 programme that has been undergoing as part of the accompanying evaluation of the Rural Development Programme 2014-2020 will be used to analyse how ÖPUL measures have contributed to the reduction of negative externalities from fertilisation and what were the costs for their implementation for farmers and society. Both measures limiting fertiliser input and mitigating negative externalities of fertilisation (e.g. buffer strips and greening measures to prevent nutrient runoff and leaching, surface near spreading of manure, etc.) will be included in the assessment. The focus will be on socio-economic impacts relating to N_2O and NH_3 emissions, N leaching to groundwater and P runoff as well as on two Austrian regions shown in Figure 18:

- The Marchfeld in the Northeast of Austria, which is characterised by low rainfall and high summer temperatures. This region is dominated by arable farming, especially cultivation of vegetables and specialist products. As shown in Figure 18 it is one of the regions with highest inputs of N fertiliser (>90 kg N/ha/a) in the country.
- A region in central Upper Austria which is delineated either according to agricultural production areas as the region between Kremsmünster and Grießkrichen or based hydrogeological conditions as the Traun-Enns-Plate. These regions are partly overlapping and will be treated as one focus region in the present case study. In addition to arable farming pig rearing is important in this region, which is reflected both in high N surpluses and high numbers of livestock density (see Figure 18).

4.3.3. Contribution to reduction of fertilisation externalities and costs of ÖPUL measures

Insights form the evaluation of the ÖPUL 2015 programme

17 ÖPUL measures have been considered relevant for the present case study as they contain restrictions on fertiliser use and/or provisions for increased nutrient retention in soil and plants. Niedermayr et al. (2019) have assessed the effects of these measures qualitatively in terms of their contribution to biodiversity, water protection, erosion prevention and soil health, reduction of greenhouse gas and ammonia emissions, carbon sequestration and animal welfare. Assuming equal assessment scales and weights for the different impact categories, effectiveness of the measures in

relation to their costs can be compared (see Table 20). The measures “Mountain grazing and herding”, “Environmentally sound management” and “Renouncement of fungicides and growth regulators in cereals” seem to be most cost-efficient, although effects of these measures are probably only caused to a minor extent by changes in fertilising practices. Among the measures primarily aimed at preventing or alleviating impacts associated with fertilisation, “Direct seeding and seeding on mulch (including strip-till)” and “Limiting yield-increasing inputs” are the most cost-efficient. On the other hand, the measures “Cultivation of mowed mountain grassland”, “Nature conservation”, “Natura-2000 agriculture” and “Preventive surface water protection” achieve improvements only under comparatively high costs. These tend to be measures for which implementation is restricted to specific areas and/or cultures.

However, the evaluation by Niedermayr et al. (2019) does not provide information on the contribution of changes in fertilisation practices to the effectiveness of a measure, nor to the consequences in terms of yield reductions implementation of ÖPUL measures implies.

Table 20: ÖPUL measures aimed at preventing or alleviating impacts associated with fertilisation and their contribution to biodiversity, water protection, erosion prevention and soil health, reduction of greenhouse gas and ammonia emissions, carbon sequestration and animal welfare. Qualitative assessment (0: no effect, 1: low effect, 2: medium effect, 3: high effect) based on Niedermayr et al. (2019). Data on payments under the ÖPUL programme from Federal Ministry Republic of Austria Agriculture, Forestry, Regions and Water Management (2022a).

Measure	Restrictions on fertiliser use	Provisions for nutrient retention	Payment [EUR/ha]	Bio-diversity	Water	Soil	N_2O & NH_3	C-Storage	Animal welfare	Total	Efficiency [(EUR/ha)/-]
Organic farming	Prohibition of N-fertiliser	Maintenance landscape-elements	250	2	3	2	1-3	1	3	11-14	18-23
Environmentally sound management	Prohibition of N-fertiliser on biodiversity areas and in greening	Maintenance of landscape elements; min. 5% biodiversity areas	60	3	1	1		1		6	10
Greening – intermediate crops	Prohibition of N-fertiliser in greening	Yearly, area-wide greening on min. 10% of arable land	152	1	1	3		1		6	25
Nature conservation	Specific obligations according to nature conservation authority	Specific obligations according to nature conservation authority	504	3	3	1-3	1-3			8-12	42-63
Preventive groundwater protection	Restrictions regarding amount and timing; mandatory nutrient balances		89		3	1		1		5	18
Mountain grazing and herding	Prohibition of N-fertiliser; spreading of manure over larger area		73	2-3	2	1			3	8-9	8-9
Greening – “Evergreen” system	Prohibition of N-fertiliser in greening	Area-wide greening on min. 85% of arable land; period without greening < 50 days	80	0-1	2	2	1-2			5-7	11-16
Limiting yield-increasing inputs	Prohibition of N-fertilisers		59	0-1	3		1-2			4-6	10-15
Erosion protection fruit/ vineyards/ hops		Yearly area-wide greening of machine tracks	205		2	3		1		6	34

Direct seeding and seeding on mulch		Greening of arable land; erosion-preventive management	59		2	2-3	1			5-6	10-12
Cultivation of mowed mountain grassland	Prohibition of N-fertiliser (except solid manure)		386	3	2	1				6	64
Surface near spreading of manure		Min. 50% of liquid farm manure spread near surface	1 EUR/m³				1-2			1-2	
Renouncement fungicides/growth regulators	Renouncement results in lower fertilisation		40	0	3		1-2			4-5	8-10
WFD-compatible agriculture	Restrictions regarding amount and timing		84	0	3	1		1		5	17
Arable areas threatened by leaching	Prohibition of N-fertilisers	Permanent green cover	430	2-3	3	3		2-3		10-12	36-43
Preventive surface water protection	Prohibition of N-fertiliser on buffer strips	Buffer strip (min. 12 m) adjacent to polluted surface rivers	990	2-3	3	3		3			
Natura 2000-Agriculture	Prohibition on mown meadows		82	0-1	3						

An exception is the measure “Surface near spreading of manure”. For this measure, reductions in N_2O and NH_3 emissions, which are the only relevant impact of this measure, have been quantified for the period 2015-2018 (see Table 21). Using the socio-economic costs of N_2O and NH_3 emissions described in Chapter 3.3.1 and 3.3.2 and listed in Table 22, benefits of implementing this measure can be monetised and compared to its costs in terms of ÖPUL payments required to motivate farmers’ participation. As can be seen from Table 21, ÖPUL payments are at the lower end of the socio-economic benefit range meaning that benefits most likely exceed socio-economic costs. It has to be noted that additional costs arising to farmers for equipment for manure spreading have not been considered in this calculation. However, as the measure is well adopted with participation rates steadily increasing, it can be assumed that ÖPUL payments constitute an adequate compensation for these additional expenses. Furthermore, if farmers would reduce their use of mineral N-fertiliser in accordance with the lower nutrient losses from manure, both private and socio-economic benefits could be increased.

Table 21: Implementation, payments, emission reductions and socio-economic benefits of the ÖPUL measure “Surface-near spreading of manure”. Data from Niedermayr et al. (2019) and Federal Ministry Republic of Austria Agriculture, Forestry, Regions and Water Management (2022a). For the calculation of socio-economic benefits see Table 22.

	2015	2016	2017	2018
Volume of manure [m³]	2 017 591	2 871 847	2 936 440	2 979 067
Payments received [Mio. EUR]	1.9			2.9
NH_3 emission reduction [t NH_3]	803	1141	1267	1273
N_2O emission reduction [t N_2O]	10	15	16	16
Socio-economic benefits from emission reduction [EUR]	-0.6 – 40.1	-0.9 – 57.1	-1.0 – 63.4	-1.0 – 63.7

Table 22: Socio-economic costs of NH_3 and N_2O emissions as described in Chapter 3.3.1 and 3.3.2. Negative costs (benefits) of NH_3 emissions are associated with aerosol cooling (see Chapter 3.3.2).

	Climate	Human health	Biodiversity
N_2O [EUR/t N_2O]	5662 – 107 876	115 – 4713	
NH_3 [EUR/t NH_3]	-3648 – 0	808 – 22 800	2000 – 11 000

Insights from the focus regions Marchfeld and Grießkirchen-Kremsmünster/Traun-Enns-plate

Two modelling studies have been conducted as part of the evaluation of the ÖPUL 2015 programme: Foldal et al. (2019) modelled the impacts of the ÖPUL measures “Organic farming” and “Environmentally sound management” as well as of two hypothetical reductions of fertiliser inputs of 15% and 25% on N_2O emissions. Their model covers five Austrian regions, including the two focus regions of the present study and takes into account regional management and local site characteristics such as climate, soil properties and plant growth. Wpa Beratende Ingenieure (2019) simulated nitrate leaching under the measure “Preventive groundwater protection” in four regions, also including both the Marchfeld and the Traun-Enns-Plate. In addition, Hölzl (2020) and Kuderna (2020) have studied effects of different ÖPUL measures on P emissions to surface waters in Upper Austria.

As shown in Figure 20 emissions of N_2O, NO_3 and NH_3 decrease in both regions with decreasing N-fertiliser inputs, except for NO_3 leaching for the measure “Environmentally sound management” in the Marchfeld. This measure is primarily aimed at creating and maintaining areas for biodiversity on farmland with restrictions on fertiliser use applying only to these areas. Therefore, it generally achieves the lowest reductions in N emissions compared to conventional farming. On the other hand, highest reductions can be observed for the measure “Organic farming”. This is not only due to stringent restrictions for fertiliser application but also a higher proportion of greening and legumes in crop rotation.

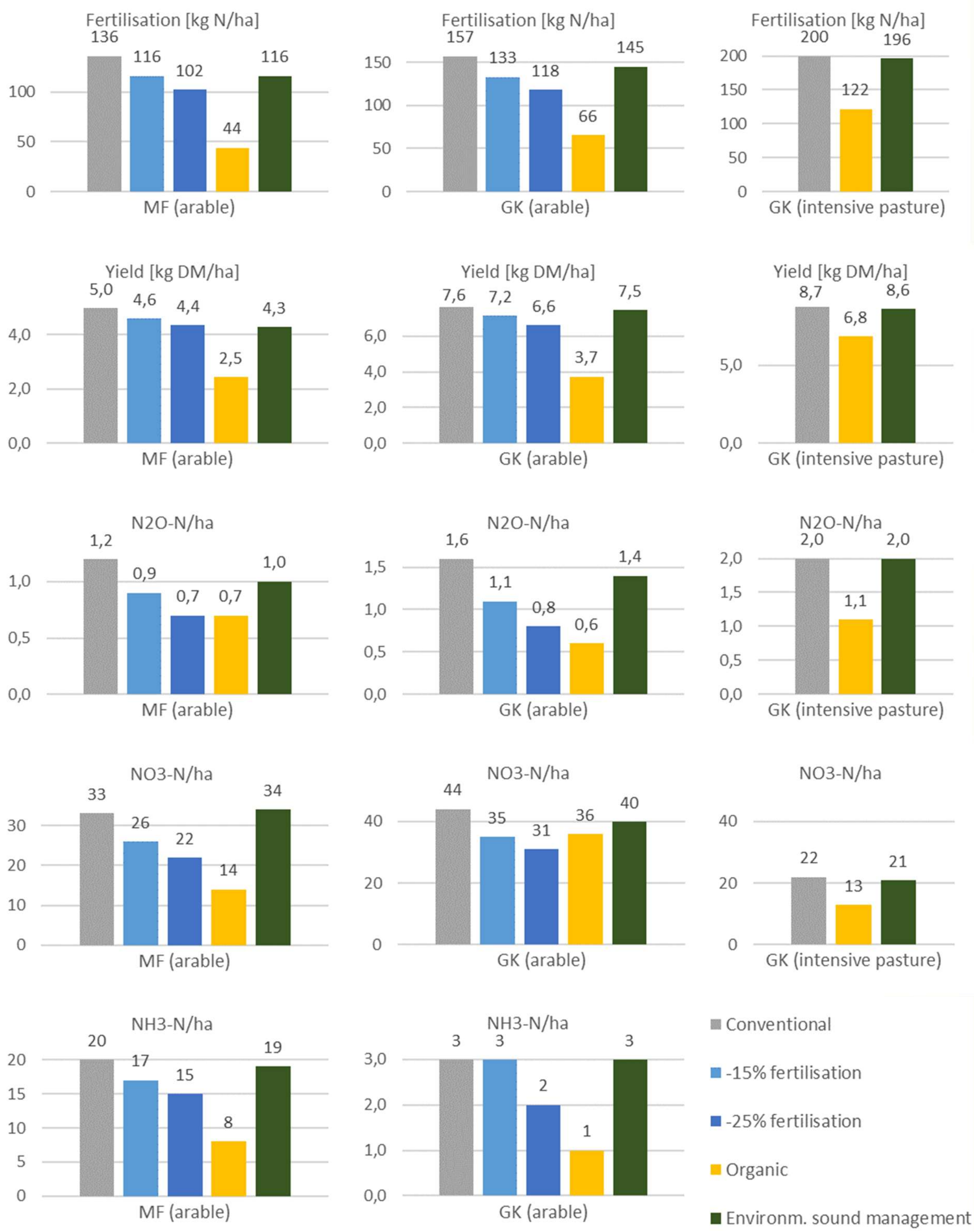

Figure 20: Fertilisation, yield and emissions of N_2O, NO_3 and NH_3 under different management practices in the two focus regions Marchfeld (MF) and Grießkirchen-Kremsmünster (GK). Mean annual values for the period 2005-2014. Data from Foldal et al. (2019). NH_3 emissions on intensive pastures in GK are negligible and therefore not depicted.

Regional differences also become evident when looking at Figure 20. In the region Grießkirchen-Kremsmünster, which is located in a higher yield area with more precipitation, fertiliser inputs, N_2O emissions and NO_3 leaching are higher than in the Marchfeld, whereas NH_3 emissions are significantly lower. Measures tend to be more effective in Grießkirchen-Kremsmünster in terms of N_2O emission reduction and in the Marchfeld regarding NO_3 leaching. For instance, the measure "Organic farming" includes reductions of N-fertiliser inputs of 68% leading to a decrease in N_2O emissions of 42% and in NO_3 emissions of 58%. In the region Grießkirchen-Kremsmünster reductions of 63% of N_2O and 18% for NO_3 are achieved at fertiliser reductions of 58% for this measure. While fertilisation intensity is the only factor causing significant changes in N_2O emission on pastures, on arable land, weather conditions also play a role. N_2O emissions in the region Grießkirchen-Kremsmünster are particularly high in hot and wet years, whereas in the Marchfeld, hot and dry years limit soil microbiological activities leading to a reduction of both plant growth and N_2O emissions (Foldal et al. 2019). Modelling results by Foldal et al. (2019) indicate that organic farmland may be less sensitive to variation in temperature and precipitation and might thus exhibit co-benefits in terms of adaptation to climate change.

Similar observations have been made by Wpa Beratende Ingenieure (2019) for the measure "Preventive groundwater protection". Reductions of N fertiliser input of 7-18% compared to conventionally managed land lead to decreases of N-losses to groundwater of 16-34% (see Figure 21). In both regions the ÖPUL measure is most effective where initial NO_3 leaching rates are high. In the Traun-Enns-Plate these are farms holding livestock in addition to arable land. This could be due to a potential underestimation of a higher soil N supply following cultivation of forage grasses. On the other hand, particularly high N-leaching in the Marchfeld occurs on land classified as medium yield areas, for which fertilisation guidelines foresee higher N-inputs than for low yield areas. The high leaching rates suggesting that classification of these areas may have to be reconsidered (Wpa Beratende Ingenieure 2019).

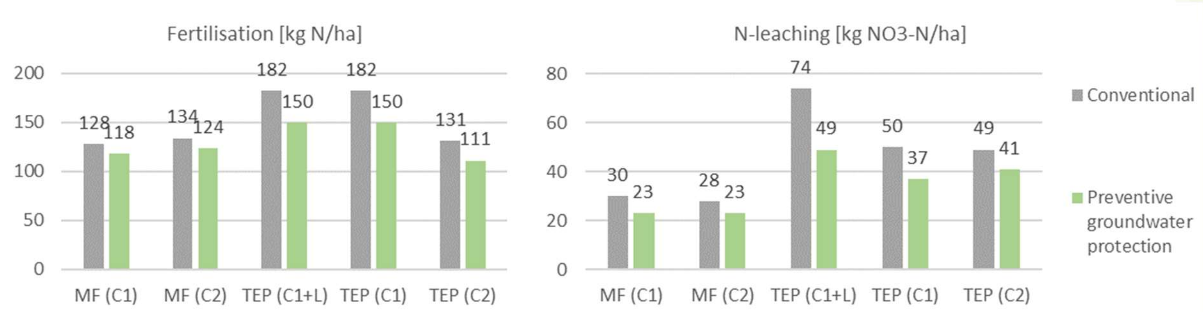

Figure 21: Impact of the ÖPUL measure "Preventive groundwater protection" on N-fertilisation and NO_3-leaching compared to conventionally managed land in the regions Marchfeld (MF) and Traun-Enns-Plate (TEP). C1 and C2 indicate different crop rotations. MF (C1): winter wheat – (geening) – sugar beet – potato – winter durum – greening – carrot – pea-spinach, MF (C2): greening – onion – winter wheat – sugar beet, TEP (C1): winter barley – greening – corn – winter wheat, TEP (C2): winter barley – greening – corn – winter wheat – soya. +L: farms with livestock and arable land. Data from Wpa Beratende Ingenieure (2019).

However, from Figure 20 it is evident that N emission reductions can only be achieved at the trade-off of decreasing yields. Emissions generally decrease to a stronger extent than yields and in a non-linear way, meaning that high effects can be achieved at comparatively low yield reductions. In the Marchfeld highest reductions of N_2O and NO_3 emissions per ton of yield dry matter are achieved at a fertiliser reduction of 25% (0.16 kg N_2O-N/kg DM and 5.1 kg NO_3-N/kg DM compared to 0.24 kg N_2O-N/kg DM and 6.6 kg NO_3-N/kg DM under conventional management). Relative NH_3 emissions are lowest under organic farming (3.3 kg NH_3-N/kg DM compared to 4.0 kg NH_3-N/kg DM under conventional management). The measure "Environmentally sound management" leads to a relative deterioration with 0.25 kg N_2O-N/kg DM, 7.9 kg NO_3-N/kg DM and 4.4 kg NH_3-N/kg DM. Reducing fertiliser input by 25% also performs best on arable land in the region Grießkirchen-Kremsmünster (0.12 kg N_2O-N/kg DM and 4.7 kg NO_3-N/kg DM compared to 0.21 kg NO_2-N/kg DM and 5.8 kg NO_3-N/kg DM under conventional management). Whereas slightly positive results can be seen for the measure "Environmentally sound management", relative NO_3 emissions increase significantly under organic farming (9.6 kg NO_3-N/kg DM). On intensive pastures both ÖPUL measures lead to relative reductions of N_2O and NO_3 emissions with higher effects achievable under "Organic farming" (0.16 kg N_2O-N/kg DM and 1.9 kg NO_3-N/kg DM compared to 0.23 kg N_2O-N/kg DM and 2.5 kg NO_3-N/kg DM under conventional management).

Nevertheless, when impacts are monetised, none of the measures is definitely able to compensate for the costs of yield reduction as shown in Table 23. In this table, socio-economic benefits of N_2O and NH_3 emission reductions are taken from Table 22; for NO_3 emission reductions benefits of 0-4 EUR/kg NO_3-N are assumed (see Chapter 3.3.3). Negative values (i.e. socio-economic costs) of NH_3 emission reductions are due to the aerosol cooling effect associated with NH_3 emissions (see Chapter 3.3.2). Benefits from avoided mineral fertiliser production following reduced fertilisation rates are calculated as EUR 0.1-4.8/kg N (combined effect of reduction of greenhouse gases, eutrophying and acidifying substances, see Chapter 3.4). On the other hand, socio-economic costs of yield reductions are reflected in an increase in land use change due to the need for expansion of agricultural area. In the best case, yield reductions can be compensated by changes in consumption patterns such as dietary habits or food waste and no land use change occurs. In the worst case, tropical forests valued at EUR 3847/ha are converted to agricultural areas (see Chapter 3.3.6). Depending on how land use changes are valued, measures yield socio-economic net-benefits or net-costs. This underlines the importance of considering and potentially counteracting yield reductions and/or their effects when designing measures for emission reduction.

Table 23: Monetised socio-economic impacts of different management practices in the two focus regions Marchfeld (MF) and Grießkirchen-Kremsmünster (GK). Negative values indicate socio-economic costs, positive values benefits. ESM: Environmentally sound management. LUC: Land use change. For the calculation of monetised impacts see Table 22, Chapter 3.3.2, Chapter 3.3.3, Chapter 3.3.6 and Chapter 3.4.

	N_2O [EUR/ha]	NO_3 [EUR/ha]	NH_3 [EUR/ha]	Av. fertiliser [EUR/ha]	LUC [EUR/ha]	Total [EUR/ha]
MF (arable)						
-15% fertilisation	1 – 21	0 – 28	-2 – 120	2 – 9	-308 – 0	-307 – 178
-25% fertilisation	3 – 36	0 – 44	-3 – 200	3 – 16	-500 – 0	-498 – 296
Organic farming	2 – 36	0 – 76	-8 – 480	9 – 43	-1962 – 0	-1961 – 634
ESM	1 – 14	-4 – 0	-1 – 40	2 – 9	-539 – 0	-537 – 59
GK (arable)						
-15% fertilisation	2 – 36	0 – 36	0	2 – 11	-231 – 0	-227 – 83
-25% fertilisation	3 – 57	0 – 52	-1 – 40	4 – 18	-500 – 0	-494 – 167
Organic farming	4 – 72	0 – 32	-1 – 80	9 – 43	-1962 – 0	-1951 – 227
ESM	1 – 14	0 – 16	0	1 – 6	-77 – 0	-75 – 36
GK (pasture)						
Organic farming	3 – 64	0 – 36	0	8 – 36	-846 – 0	-835 – 136
ESM	0	0 – 4	0	0 – 2	-53 – 0	-53 – 6

Compared to the range of benefits achievable, ÖPUL payments of EUR 250/ha for "Organic farming" and EUR 60/ha for "Environmentally sound management" appear high. However, it has to be born in mind that these measures include benefits not directly linked to fertiliser use and therefore out of the scope of the present study such as reduced leaching and runoff of pesticides or protection of biodiversity (see Table 20). From a farmers' perspective, ÖPUL payments seem to provide adequate compensation for the costs arising from the implementation as both measures are among the most frequently adopted ones in the ÖPUL programme.

For the measure "Preventive groundwater protection" impacts on yield reduction have not been reported. Instead, farmers were asked for their assessment. 76% of respondents consider compensation payments for the measure "Preventive groundwater protection" (rather) insufficient, although only a slight majority (55%) report negative impacts on their operating results. Yield reductions and reductions in crop quality due to restrictions of fertilisation are the most frequently cited reasons for those losses. Temporal restrictions on fertilisation and documentation duties constitute other repeatedly mentioned hurdles for participation. These factors are particularly important for farms holding livestock, as temporal fertilisation restrictions require manure storage facilities and documentation is associated with a higher effort. Consequently, participation in the Marchfeld is with 83% of arable area in 2017 considerably higher than in the Traun-Enns-Plate (49%). In light of these hurdles, representatives of the regional chambers of agriculture advocate for weakening restrictions on fertilisation and higher premiums to increase participation. Water suppliers on the other hand would prefer a more stringent formulation of the measure (Wpa Beratende Ingenieure 2019).

Agricultural P emissions have only been studied for Upper Austria. Prior to the ÖPUL 2015 programme threshold values for river PO_4-P concentrations were exceeded in all but one of the river basins in the region, with exceedance rates of 1.2-3.6. Erosion from agricultural land is the dominant source of P emissions. Through the implementation of ÖPUL measures, P emissions could be reduced by 6-12%, which is insufficient in respect to threshold values. The majority of the reduction can be attributed to prescriptions for greening and/or direct seeding and seeding on mulch (4-12%), while the measure "Organic farming" also contributes with up to 2% (Kuderna 2020). However, as these numbers refer to impacts in the whole region, higher adoption rates of greening measures (e.g. in the range of 20-30% for "Greening – intermediate crops") than of "Organic farming" (typically below 10%) have to be taken into account (Wpa Beratende Ingenieure 2019). Promotion of participation in ÖPUL measures could thus increase their impact. Moreover, soils in the region, especially on pastures, frequently exhibit a P-deficit (Hölzl 2020). Adoption of ÖPUL measures aimed at erosion reduction could therefore even have positive effects on yields. This impact remains to be quantified though.

4.3.4. Case study conclusions

Overall, evaluation of the ÖPUL 2015 programme shows that measures are effective in reducing nutrient emissions to water and atmosphere, protecting soil health and biodiversity and promoting animal welfare. Yet, with respect to impacts of fertilising practices it remains unclear whether the amount of premiums granted to farmers under the programme are justified.

On the one hand, trade-offs with yield reduction have to be considered. Depending on the assumptions taken with respect to the extent of land use changes required due to lower productivity of agricultural land and the type of land converted, implementation of ÖPUL measures could even result in net socio-economic costs of EUR 1951/ha (not including premiums paid to farmers). Therefore, the ÖPUL programme should be accompanied by measures to alleviate the effects of yield reductions such as the promotion of healthy diets and reduction of food waste. Furthermore, focus should be on regions, crops and measures for which highest emission reductions per yield can be achieved. Results from the two focus regions show that this is typically the case for moderate reductions of N fertiliser inputs and where initial emissions are particularly high. Measures aimed at erosion prevention could even reduce

P emissions to surface water and improve soil P balances simultaneously. Similarly, advanced fertiliser application techniques such as promoted in the ÖPUL measure "Surface near spreading of manure" or novel, low-emission bio-based fertilisers could reduce nutrient emissions without compromising yields.

Furthermore, results of the effectivity of ÖPUL measures are highly dependent on assumptions made with respect to socio-economic costs of NH_3 emissions, i.e. whether positive effects on climate change due to aerosol cooling are valued higher than negative health costs associated with PM formation.

On the other hand, a balance between stringency of the measures to maximise their efficiency and farmers' acceptance to promote widespread adoption has to be found. As shown in the survey by Wpa Beratende Ingenieure (2019), farmers tend to regard ÖPUL premiums as insufficient even if they do not always experience negative impacts on their operating results. Restrictions on fertilisation are viewed particularly critical. In contrast, water suppliers call for stricter regulations in ÖPUL measures, but also for higher participation rates than are currently reached. An approach could be staggered premiums with higher payments for more stringent adoption, accompanied by targeted training measures, informing farmers about their real implementation costs and providing guidance on how to avoid yield losses when implementing the measure.

Finally, it has to be noted that impacts on biodiversity, which are an important component of many ÖPUL measures but usually not directly associated with provisions related to fertilising practices, have not been considered in the monetisation of impacts. Inclusion of these impacts would likely improve the socio-economic cost-benefit ratio of ÖPUL measures.

4.4. Stakeholder perspectives in the case study regions

To complement the analysis in Chapter 4.1 to Chapter 4.3, stakeholder perspectives from different case study regions and sectors (industry, public administration and research) were collected. More precisely, the following individuals and organisations were involved: The University of Latvia and GreenBack Sp. Z o.o., an environmental consultancy, in the Baltic Sea region and the Austrian Environmental Agency in Austria. Interviews focussed on the impacts of bio-based fertilisers, which have not been extensively covered in the literature to date yet. They were based on the questionnaire shown in Table 24 and adapted to the respective respondent's expertise and experience.

Table 24: Questionnaire used as basis for stakeholder interviews in the three case-study regions.

How important are the following impacts of usage of (a) mineral fertilisers and untreated manure and (b) bio-based fertilisers in agriculture in your region?

	Not at all important	Low importance	Neutral	Important	Very important	No opinion
Environmental factors/ aspects						
Recirculation of waste materials						
Increased degradation of organic matter in agricultural soil						
Soil pollution (Cd, Cu, Zn, etc.)						
Acidification of agricultural soil						
Aquatic eutrophication (N and P runoff)						
NO_3 to groundwater						
Increased N_2O emissions from waterbodies						

Increased organic matter in agricultural soil						
Terrestrial eutrophication						
Aerosol formation induced by NH_3 and NO_x (atmospheric cooling, increased plant growth)						
Particular matter formation						
Tropospheric ozone formation induced by N_2O						
Decreased plant production induced by O_3						
Stratospheric ozone depletion						
Global warming (fertiliser production, N_2O soil emissions, O_3)						
Biodiversity loss due to acidification and eutrophication						
Increase in occurrence of parasitic and infectious diseases due to eutrophication						
Release of toxic compounds due to soil acidification						
Enhanced C sequestration due to terr. eutrophication						
Contaminants released under enhanced NO_3 concentrations in groundwater						
Avoidance of land use change due to intensification of production						
Economic factors/ aspects						
Creation of labour in manufacturing sector						
Change in crop quality						
Maintenance of labour in agriculture						
Contribution to manufacturing gross value added (GVA)						
Contribution to agricultural gross value added (GVA)						
Damage to buildings and materials due to acidification						
Losses in tourism and leisure sector due to eutrophication						
Losses in fishing sector due to eutrophication						
Losses in real estate value due to eutrophication						
Social and other factors/ aspects						
Risk of major accidents at fertiliser plants						
Contribution to poor work-life balance (fertilizer industry)						
Contribution to work-related stress (fertilizer industry)						
Contribution to employee empowerment (fertiliser industry)						
Contribution to typical and regular work hours (fertilizer industry)						
Contribution to secure working conditions (fertilizer industry)						

Contribution to gender pay gap (fertilizer industry)						
Geopolitical dependence on countries with phosphorus (P) & potassium (K) reserves						
Contribution to food security						
Risk of major accidents at phosphogypsum stacks						
Other impacts						
Please name						

Experts from all three institutions consider the increase of organic matter in agricultural soil and changes in crop quality as important factors for the use of BBFs. For Austria, better maintenance of soil health compared to conventional fertilising practises is particularly pointed out. From an economic perspective, BBFs are regarded to benefit both GVA and employment, potentially even more than mineral fertilisers. Nevertheless, like mineral fertilisers and untreated manure, BBFs are seen as contributors to terrestrial and aquatic eutrophication, acidification of agricultural soil and NO_3 emissions to groundwater. This is especially the case for the Baltic Sea region, whereas the expert from the Austrian Environmental Agency considers impacts of BBFs lower than those of conventional fertilisers and sees significant impacts of BBFs only in terms of terrestrial eutrophication.

5. DISCUSSION AND CONCLUSIONS

Table 25 provides an overview of all socio-economic impacts of current fertilising practices described in Chapter 3 and their costs and benefits in monetary terms, where available. It is evident that a conclusive assessment of the true costs of current fertilising practices is not possible under the current state of knowledge. First, of the 60 impacts identified only 24 could be monetised and among those cost estimate ranges covering several orders of magnitude are not uncommon. Moreover, narrow cost estimate ranges often point to few studies having attempted to monetise a particular impact rather than a higher degree of certainty. Second, it has to be born in mind that even though all impacts are expressed in monetary units, calculation with different methods may hamper their comparability. However, for the impacts shown in Table 25 differences between calculation approaches seem at least not to be larger than uncertainties within a method. Finally, impacts may partly overlap. For instance, the benefits of increased yields due to fertilisation can be either assessed as the reduced amount of agricultural area needed to feed a certain population, or as the amount of additional people fed on a given amount of land. Due to the higher valuation of a life year (EUR 57 000 – 138 700, see Chapter 3.6.1) than of a ha of natural land (up to EUR 3847 per year, see Chapter 3.3.6) estimated benefits reach from EUR <1 billion to EUR >30 000 billion (see Table 25). On the other hand, due to the cascading effects in the nitrogen cycle, a single molecule of reactive nitrogen released can contribute to multiple impacts, which are therefore additive rather than complementary.

Table 25: Overview of costs and benefits of current fertilising practices. Summary of impacts described in Chapter 3.

Impact	Benefits (billion EUR)	Costs (billion EUR)	Chapter
Economic impacts			
Contribution to turnover of the manufacturing sector	10		3.1
Creation of labour in manufacturing sector	Not quantified		3.1
Contribution of fertiliser use to agricultural GVA	7		3.2
Maintenance of agricultural employment	Not quantified		3.2
Environmental impacts			
Global warming effects of fertiliser production		1-64	3.4
Eutrophication effects of fertiliser production		<1	3.4
Acidification effects of fertiliser production		<1	3.4
Leaching from phosphogypsum stacks		40-49	3.4
Effects of direct N_2O emissions on climate		3-105	3.3.1
Effects of direct N_2O emissions on health (ozone)		<1	3.3.1
Impacts on human health of NH_3 and NO_2 emissions and related substances		3-139	3.3.2
Aerosol impact on climate (NH_3 and NO_x induced)	<1-11		3.3.2
Increase of plant productivity due to aerosols (NH_3 and NO_x induced)	Not quantified		3.3.2
Stratospheric O_3 induced amplification of global warming		Not quantified	3.3.2
Tropospheric O_3 induced amplification of global warming		Not quantified	3.3.2
O_3 induced crop damage		<1	3.3.2
Reduced carbon sequestration due to O_3 plant damage		<1-11	3.3.2
Biodiversity loss due to terrestrial acidification and eutrophication		6-374	3.3.2
Increase in occurrence of parasitic and infectious diseases due to terrestrial eutrophication		Not quantified	3.3.2
Increase in allergenic pollen production due to terrestrial eutrophication		Not quantified	3.3.2
Release of toxic compounds due to soil acidification		Not quantified	3.3.2
Damage to buildings and materials due to acidification		<1	3.3.2
Indirect N_2O emissions due to NH_3 and NO_x emissions		<1-4	3.3.2
Enhanced C sequestration due to N deposition in natural ecosystems	<1-152		3.3.2
Enhanced CH_4 emissions due to N deposition in natural ecosystems		Not quantified	3.3.2
Increased albedo of vegetation due to increased N deposition	Not quantified		3.3.2
Increased N leaching due to N deposition		Not quantified	3.3.2
Increased N_2O emissions due to increased N leaching		Not quantified	3.3.2
Health risks of NO_3 groundwater contamination		<1-7	3.3.3
Health risks/treatment costs of other groundwater contaminants released under enhanced NO_3 concentrations		Not quantified	3.3.3
Water treatment costs for NO_3 removal from drinking water		Not quantified	3.3.3
Losses in tourism and leisure sector due to aquatic eutrophication		Not quantified	3.3.4
Indirect N_2O emissions from N runoff and leaching		<1-16	3.3.4
Losses in fishing sector due to aquatic eutrophication		Not quantified	3.3.4
Losses in real estate values due to aquatic eutrophication		Not quantified	3.3.4
Increased risk of infectious diseases due to aquatic eutrophication		Not quantified	3.3.4
Increased drinking water purification costs due to aquatic eutrophication		Not quantified	3.3.4
Increased CH_4 emissions due to aquatic eutrophication		Not quantified	3.3.4
Biodiversity losses due to marine eutrophication		6-26	3.3.4
Biodiversity losses due to freshwater eutrophication		89-98	3.3.4
Increase in soil organic matter on agricultural land	Not quantified		3.3.5
Liming requirement due to acidification of agricultural land		Not quantified	3.3.5

Risk of pollutant leaching from agricultural land after abandonment		Not quantified	3.3.5
Health risks (Zn, Cu) due to manure application on agricultural land		<1-1254	3.3.5
Cd contamination of agricultural land due to mineral P fertilisers		2-4	3.3.5
Health risks of Zn, Cu, Ni and U in mineral P fertiliser		Not quantified	3.3.5
Risk to soil organisms due to Cd, Cu and Zn in fertilisers		Not quantified	3.3.5
Avoidance of land use change	<1-309		3.3.6
Changes in food quality	Not quantified	Not quantified	3.3.7
Odour nuisance during manure management and spreading		Not quantified	3.3.8
Social and other impacts			
Risk of major accidents during fertiliser production		Not quantified	3.5.1
Environmental hazard of phosphogypsum stacks		Not quantified	3.5.1
Contribution to closing the gender pay gap		Not quantified	3.5.2
Avoidance of precarious working conditions	Not quantified		3.5.2
Contribution to work-life-balance and mental wellbeing		Not quantified	3.5.2
Contribution to physical well-being of workers	Not quantified		3.5.2
Contribution of employee education	Not quantified		3.5.2
Contribution to empowerment of employees	Not quantified		3.5.2
Contribution to food security	14 188-34 106		3.6.1
Economic and political dependence on third countries		Not quantified	3.6.2

Nevertheless, from Table 25 it is clear that the benefits of increased yields could only be achieved at the costs of various, mainly environmental trade-offs that entail further consequences for economy and society and as shown in the Baltic Sea case study (Chapter 4.1), can have far-reaching impacts into the future. Table 25 also shows that (both positive and negative) impacts of fertilisation mainly occur in the use phase. Compared to a global assessment of the contribution of anthropogenic N_r emissions to the exceedance of ecosystem and health thresholds (Erisman et al. 2013, see Table 26), terrestrial biodiversity losses seem to be more relevant in the context of European N fertilising practices, whereas health risks due to NO_3 contamination of drinking water seem to play a smaller role. Similarly, marginal social costs of NO_3 pollution caused by N fertilisation in Minnesota as assessed by Keeler et al. (2016) are with USD 0-0.5/kg N below those associated with the emissions of NH_3 (USD 0.1-50/kg N), NO_x (0.001-0.5/kg N) and N_2O (USD 0.1-0.5/kg N). As the study by Erisman et al. (2013) does not include an element of monetisation, this partly reflects differences in the societal valuation of impacts.

Table 26: Exceedance of global ecological and health thresholds by anthropogenic N_r emissions. Data from Erisman et al. (2013). When comparing to Table 25, note the different assessment scale (global, all sectors, N emissions) and the fact that no element of valuation is contained in the assessment.

	Exceedance of ecological/ health threshold for N emissions	Contribution of ant. N_r emissions to threshold exceedance	**Exceedance of ecological/ health threshold caused by ant. N_r**
NO_3 & NO_2 intake by humans	70%	80%	**56%**
Coastal dead zones	80%	50%	**40%**
Air pollution (human health)	60%	22%	**13%**
Stratospheric ozone depletion	20%	40%	**8%**
Terrestrial biodiversity loss	50%	15%	**8%**
Freshwater pollution	10%	40%	**4%**
Air pollution (crop loss)	4%	50%	**2%**
Climate change	20%	-15%	

Although neither on a European level nor for the three case studies final conclusions on whether benefits of current fertilising practices outweigh costs or vice versa can be made, the assessment points out that there is at least room for improvement in the cost-benefit ratio of current fertilising practices.

In the Baltic Sea region, for instance, biological remediation of legacy nutrient loads through mussel farming, targeted fishing or harvesting of algae could even open up new business opportunities in a circular economy. Provided the harvested biomass proves suitable to process into effective and marketable BBFs impacts of fertiliser production may be reduced simultaneously. However, it has to be born in mind that if remediation activities are to be conducted on a scale sustaining the integrity of the marine environment, to achieve visible effects within a reasonable timeframe, they have to be integrated into a holistic approach including measures targeting emission reduction at source.

Regarding the Austrian agri-environmental programme, it has been shown that while most ÖPUL measures have proven effective, efficiency of the payments could be increased if focused especially on areas where emission reductions are achievable at lowest yield losses, i.e., typically areas, where initial emissions are particularly high. Accompanying policies targeting shifts towards a healthy diet and reduction of food waste could also contribute to offset the costs of yield reductions. Furthermore, if the voluntary character of the programme is to be kept, it has to be born in mind that the overall effect of a measure is dependent on both its stringency and participation rates, reflecting its acceptance by the farmers.

For Flanders, it could be demonstrated that when taking external health costs of ammonia emissions into account, fertilisation yields socio-economic benefits. However, the current practice of fertilising with minimally processed manure and synthetic fertilisers leads to a socio-economic suboptimal welfare. A higher share of synthetic fertilisers and recovered nutrients from manure in the fertiliser mix would be preferable. The potential of BBFs to improve the socio-economic cost-benefit ratio of fertilising practices has been confirmed by the three interviewed experts in the Baltic Sea region and Austria. Yet, their application is not free from environmental trade-offs either. More emphasis regarding measurement, simulation and assessment of emissions from various fertilising products is therefore needed and targeted solutions to regional circumstances and problems are likely to be necessary.

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