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# Climate mitigation co-benefits from sustainable nutrient management in agriculture

## Incentives and opportunities

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# Abstract

Nitrogen management policies introduced in the past decades by some OECD countries have succeeded in reducing excess nitrogen use by farmers, but half of global mineral fertiliser use is still lost for crops. While about half of OECD countries have nutrient surpluses of between 25-50 kg N per hectare, a smaller number of countries are still having surpluses of more than 100 kg N per hectare. Since the production and use of mineral fertilisers have a large greenhouse gas footprint and to achieve the deep reductions in emissions as the Paris Agreement aims for, nitrogen management policies could be reinforced and pursued more systematically. The paper identifies significant reduction potential by eliminating the excess use of nitrogen fertilisers and improving efficiency in the use of manure-nitrogen, which could be obtained with a redesign of nitrogen management policies and schemes for public financial support. To underpin such measures a tax on the nitrogen surplus at farm level could play a vital role. Based on the available estimates of environmental externalities of nitrogen, the paper identifies an average rate of EUR 1-2 as a suitable starting point for a tax or penalty on the surplus application of nitrogen. The paper also explores the opportunities for sustainable nutrient management in agriculture with climate mitigation benefits relating to nitrous oxides in particular.

**Keywords**: climate change mitigation, nitrogen pollution, agricultural fertilisers, environmental taxation, manure.

**JEL codes:** D62, H23, H87, O13, P52, Q15, Q24, Q51, Q55, Q58

# Résumé

Les politiques de gestion de l'azote introduites au cours des dernières décennies par certains pays de l'OCDE ont réussi à réduire l'utilisation excessive d'azote par les agriculteurs, mais la moitié de l'utilisation mondiale d'engrais minéraux est toujours perdue pour les cultures. Alors qu'environ la moitié des pays de l'OCDE ont des excédents d'éléments nutritifs compris entre 25 et 50 kg N par hectare, un plus petit nombre de pays ont encore des excédents de plus de 100 kg N par hectare. Étant donné que la production et l'utilisation d'engrais minéraux ont une forte empreinte de gaz à effet de serre et pour atteindre les fortes réductions d'émissions visées par l'Accord de Paris, les politiques de gestion de l'azote pourraient être renforcées et poursuivies de manière plus systématique. Le document identifie un potentiel de réduction significatif en éliminant l'utilisation excessive d'engrais azotés et en améliorant l'efficacité dans l'utilisation de l'azote du fumier, ce qui pourrait être obtenu avec une refonte des politiques de gestion de l'azote et des programmes de soutien financier public. Pour étayer ces mesures, une taxe sur l'excédent d'azote au niveau de l'exploitation pourrait jouer un rôle essentiel. Sur la base des estimations disponibles des externalités environnementales de l'azote, le document identifie un taux moyen de 1 à 2 euros comme point de départ approprié pour une taxe ou une pénalité sur l'épandage excédentaire d'azote. Le document explore également les opportunités de gestion durable des nutriments en agriculture avec bénéfices d'atténuation du changement climatique liés aux oxydes nitreux en particulier.

**Mots clefs**: atténuation du changement climatique, pollution azotée, engrais agricoles, fiscalité environnementale, fumier.

**Classification JEL** : D62, H23, H87, O13, P52, Q15, Q24, Q51, Q55, Q58

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# Executive summary

Efforts over the past decades by several OECD countries to reduce nitrogen (N) leaching to water bodies by substituting the use of mineral fertilisers with manure nutrients provide an important experience, relevant for achieving deep reductions in greenhouse gas (GHG) emissions from agriculture, as expected under the 2015 Paris Agreement to the UN Framework Convention on Climate Change. As the Intergovernmental Panel on Climate Change (IPCC) noted, in 2019, half of global mineral fertiliser use is lost to crops. With reductions in fertiliser demand targeted where agricultural producer emissions are highest, the paper shows that there could be a global GHG mitigation potential corresponding to 20% of global agricultural emissions when manure-N, where available, is used more systematically in OECD countries as a substitute for mineral fertilisers.

The objective of this paper is hence to explore the interface of nitrogen management and climate change mitigation. Specifically the paper considers how nitrous oxide and methane emissions from agricultural nutrient management could be reduced. These are potent GHGs with a global warming potential orders of magnitude higher than carbon dioxide, even though methane is a short-lived GHG. The paper takes stock of the nutrient surpluses, their drivers and some of the key abatement measures available, which leads to a discussion of possible policy instruments and the further co-benefits available.

While about half of OECD countries have nutrient surpluses of between 25-50 kg N per ha, a smaller number of countries have surpluses of more than 100 kg N per ha. However, these are national figures that may conceal hotspots at regional and local level, notably in the middle-sized and larger countries. The drivers of surpluses are numerous, and include as well the increasing specialisation of agriculture with large livestock facilities separated from arable land and facing considerable costs for disposing appropriately of their manure, as the propensity of arable farmers to risk-minimise on yields by topping up their mineral fertiliser doses. While there is no simple correlation between the public financial support to farmers and the nitrogen surplus, the highest surpluses can frequently be observed in those countries where support is most generous and distorting.

The hierarchy of abatement measures range from containment and capturing of manure nutrients, to employment of novel cleaner technologies and structural measures in the agricultural sector to reduce nutrient volumes. To obtain a high utilisation rate of manure nutrients, the seasonal timing of their spreading is essential, for which appropriate and sealed storage capacity must be available. To capture all manure nutrients for crop needs, the appropriate spreading technologies need to be mobilised, while system optimisations can be achieved via novel cleaner technologies enabling distributed nutrient management. One example is the processing in biogas units that increase the short-run nutrient availability in manure, while offering opportunities for integration with intermittent energy supply. It may be developed into innovations at system level when integrated into full-scale bio-refineries, with higher value spin-offs in terms of protein or jet-fuel supply for example, although these are still experimental. Other trajectories of structural innovations involve organic farming and conservation agriculture, complemented by regulations on animal densities.

Life cycle analyses show that the carbon footprints of mineral fertilisers stem as well from their application to land as from their production phase. While the carbon pricing scheme of the European Union (Emissions

Trading System; EU ETS) has led to notable reductions in emissions of fertiliser producers, other world regions continue to produce mineral fertilisers with a carbon footprint 2-3 times higher. The full carbon footprint of mineral fertiliser-N thus ranges from 10-20 kg GHG emissions per kg N globally. However, organic manure-N cannot substitute inorganic N from mineral fertilisers 1:1, as the mineralisation processes in soils upon application of manure-N proceed over several years, depending on soil properties and the local climate. Regulators can however provide guidance to farmers by defining utilisation rates for various types of manure. To be environmentally and economically sound, such utilisation rates should take into account not only the immediate first-year availability of nitrogen in manure, but as well the availability in subsequent years. Although mineralisation of solid dung proceeds more slowly than of slurry, the inertias in land use will frequently imply the availability of nutrients from the applications during former cultivation years. Book-keeping of nutrient applications and mandatory reporting is thus essential for supervision and control.

Besides permits for large livestock operations, current policy approaches remain oriented towards voluntary efforts among farmers in making use of manure, stimulated by public financial support, although in Europe the designation of nitrate vulnerable zones will make the national Codes of Good Agricultural Practice mandatory to farmers. While providing an overview of the regulatory approaches in the EU and United States, the paper takes interest in experiences with economic instruments in terms of taxes on mineral fertilisers. It is challenging for economic analyses to capture the finer elements of manure management and farmer behaviour, making regulations frequently appear overtly expensive, e.g. by reducing livestock or yields. That there are some win-win options available has however become clear from the experiences in countries that have introduced taxes or charges on mineral fertilisers, as documented in ex-post econometric studies, reflecting some inefficiencies at play in the agricultural sector.

Abatement costs of around EUR 0.1-0.6 per kg N are contrasted with findings of studies on the external costs on nitrogen surpluses that, while being heterogeneous due to the complexities of catchments and the site-specific pathways, can range up to EUR 10 per kg N. Although accounting for the externalities is challenging and warrants further research, there are apparent co-benefits within reach from considering implications to the wider nitrogen cycle of abatement. In contrast, using the social cost of carbon as the only yardstick for mitigation measures relating to the GHGs of the nitrogen cycle is not likely to be sufficient in identifying suitable measures. Based on the available estimates of the external costs of nitrogen, as derived from a panel of catchments, the paper identifies an average rate of EUR 1-2 as a suitable starting point for a tax or penalty on the surplus application of nitrogen. Still, further research is advisable for designing schemes adjusted to national circumstances.

# 1 Introduction

With a global population expected to reach 9 billion people by mid-century there is a need for food production to continue expanding to keep up with and satisfy demand. Over the past fifty years, worldwide fertiliser use has increased four-fold, for nitrogen even with a factor of seven, supporting decisively agricultural productivity. Still, fertiliser use is also associated with undesirable side-effects on the local and global environment. In relation to the climate crisis about 4.5% of greenhouse gas (GHG) emissions relate to the use of mineral and organic fertilisers by agriculture. Moreover, global data suggests, that of the fertiliser nitrogen applied, crops utilise less than half, with a substantial share lost through runoff, leaching or volatilisation (IPCC, 2019). Erisman et al. (2008) estimate that less than 20% of the nitrogen used in global agriculture is in the end consumed with crop, dairy and meat products by humans.

With the 2015 Paris Agreement to the UN Framework Convention on Climate Change (UNFCCC), a consensus has been reached to aim for keeping the increase in the global average temperatures to well below 2 degrees Celsius above pre-industrial levels and to undertake rapid reductions in emissions. Moreover, several OECD countries and regions have since the 1990s put domestic policies in place to manage fertilisers with the objective of reducing nutrient losses to limit the associated environmental implications. It is timely to revisit the opportunities and measures for simultaneously closing the nutrient cycle while limiting emissions of GHGs from agriculture.

Three GHGs are relevant in this context; while carbon dioxide ($CO_2$) is emitted mainly from fossil fuels used for production and transport of fertilisers, there are significant amounts of nitrous oxides ($N_2O$) released from both land use practices, manure and upstream industrial processes, as well as methane ($CH_4$) emissions from livestock and the manure generated. Both $N_2O$ and methane are powerful greenhouse gases, with a potency per unit of weight many times that of carbon dioxide. Using the global warming potential for a 100 years' time horizon, one unit of $N_2O$ corresponds to 298 units of $CO_2$, while methane, despite its shorter lifetime, corresponds to 25 units of $CO_2$ according to UNFCCC's methodology prescribed to Annex-I countries. On a 20-years' time horizon, methane is 84 times as potent in terms of global warming as $CO_2$. In developed countries, mineral and organic fertilisers thus account for more than 40% of total GHG emissions from agriculture (WRI and WBCSD, 2014), with the specific mix depending on national management practices, e.g. as related to manure.

A conventional distinction is between mechanical and non-mechanical sources of GHGs (Table 1.1). Agriculture relies on biological systems, whose emissions occur through more complex mechanisms than the emissions from the mechanical equipment used on farmland. Non-mechanical emissions are shaped by climatic and soil conditions (e.g. decomposition) or the burning of crop residues, often with complex patterns of nitrogen (N) and carbon (C) flows through farms. While $N_2O$ emissions result from nitrification and denitrification processes, methane emissions stem from methanogenesis under anaerobic conditions in soils and manure storage, enteric fermentation and the incomplete combustion of organic matter. While it is difficult to quantify exactly the GHG emissions from agriculture, it is safe to observe that non-mechanical sources are by far the most important ones. Enteric fermentation (methane cow burps) and soils ($N_2O$) are the most significant sources at farm level, but there are important upstream sources in the production of farm inputs, addressed in Chapter 4 on the carbon footprint of mineral fertilisers.

At farm level the significance of the various GHGs differ with the type of farm, its management practices and the specific natural factors, such as land cover; topography and hydrology; soil temperature, moisture, organic content and composition; crop or livestock specialisation and land use practices.

## Table 1.1. Sources of greenhouse gas emissions from agriculture

| Mechanical[1] | GHG | Non-mechanical | GHG |
|---|---|---|---|
| Purchased electricity | $CO_2$, $CH_4$, $N_2O$ | Drainage and tillage of soils | $CO_2$, $CH_4$, $N_2O$ |
| Mobile machinery (e.g. tilling, sowing, harvesting, and transport and fishing vessels) | $CO_2$, $CH_4$, $N_2O$ | Addition of synthetic fertilisers, livestock waste and crop residues to soils | $CO_2$, $CH_4$, $N_2O$ |
| Stationary machinery (e.g. milling and irrigation equipment) | $CO_2$, $CH_4$, $N_2O$ | Addition of urea and lime to soils | $CO_2$ |
| Refrigeration and air-conditioning equipment[2] | HFC, PFC | Enteric fermentation | $CH_4$ |
| | | Rice cultivation | $CH_4$ |
| | | Manure management | $CH_4$, $N_2O$ |
| | | Land use change | $CO_2$, $CH_4$, $N_2O$ |
| | | Open burning of savannahs and of crop residues left on fields | $CO_2$, $CH_4$, $N_2O$ |
| | | Managed woodland (e.g. tree strips, timber belts) | $CO_2$ |
| | | Composting of organic wastes | $CH_4$ |
| | | Oxidation of horticultural growing media (e.g. peat) | $CO_2$ |

Notes: 1. Mechanical includes building heating.
2. HFC = hydrofluorocarbons; PFC = perfluorocarbons.
Source: WRI and WBCSD, 2014.

GHGs comprise the following main sources at farm level as highlighted in Table 1.1:

- Methane emissions from the storage of animal manure
- Methane emissions from manure of grazing animals
- $N_2O$ emissions from the application of animal manure to crops
- $N_2O$ emissions from manure of grazing animals
- $N_2O$ emissions from the application of other organic nutrients (compost, biomass, sewage sludge etc.) to crops
- $N_2O$ emissions from the application of mineral fertiliser to crops
- $N_2O$ emissions indirectly via leaching and volatilisation
- N-fixation of legumes and incorporation of crop residues into soils affecting $N_2O$
- N-mineralisation associated with loss of soil organic matter from land use changes affecting $N_2O$

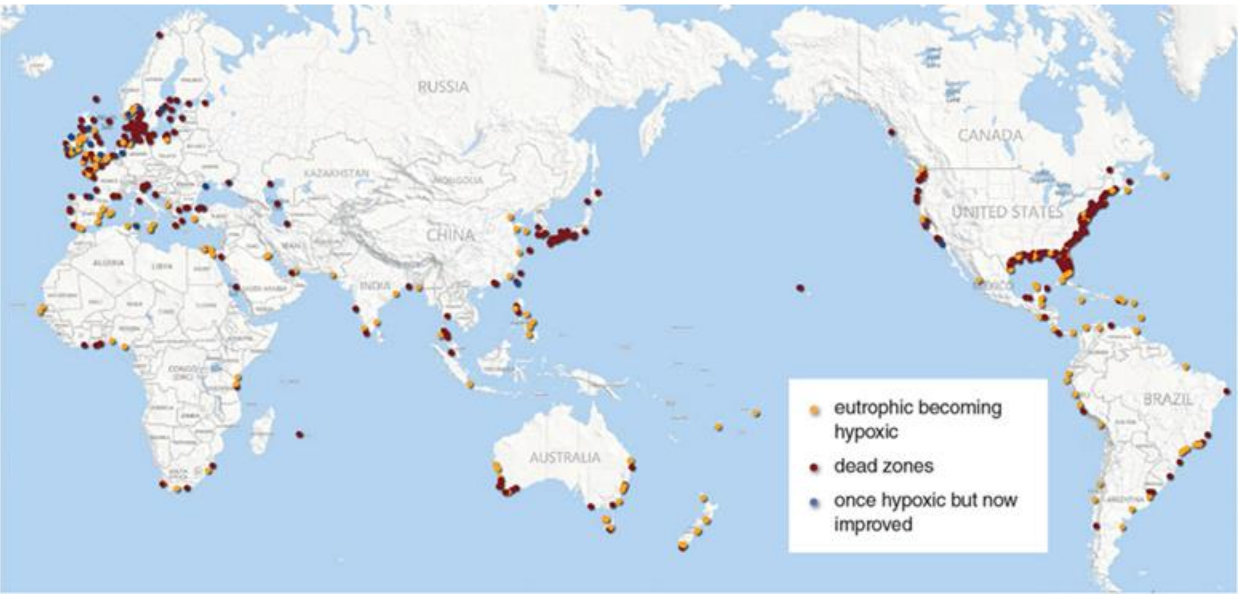
In 2007-16 agriculture accounted for around 78% of $N_2O$ emissions and 39% of methane emissions from human activities worldwide, or 12% of the net total of anthropogenic GHG emissions[1] (IPCC, 2019). The present paper focuses on joint reductions of GHGs and nitrogen and thus mainly on $N_2O$ emissions from inorganic and organic (manure) fertilisers, while considering also methane emissions from these sources. The paper focuses on the agricultural sector, recognising that cost-effective environmental policy requires considering all sources of nitrogen (see OECD, 2018a).

[1] Excluding $CO_2$, for which no global data is available for agricultural emissions (IPCC, 2019).

# 2 Structural drivers of nutrient management challenges.

Challenges with eutrophication and hypoxia in coastal waters have been acknowledged since the 1950s and are widespread in OECD countries, frequently with nitrogen as the controlling factor (Vollenweider, 1968). From initial occurrences in the Adriatic Sea and Northern Europe the phenomenon has multiplied, and since the late 1990s has become common in all of Europe as well as in North America and East Asia (Figure 2.1). It was gradually understood that use of mineral and organic fertilisers in agriculture had become a dominant source of nitrogen losses to air, water and soil, triggering impacts on the terrestrial environment and biodiversity too. From the Third Ministerial Conference on Environment and Health, organised by the World Health Organization in 1999, the human health impacts of nitrogen contaminations of drinking water gradually came into focus, as well as the possible human health impacts from airborne nitrogen (ammonia). Intake of nitrates via water may increase occurrence of cancers and early mortality (Schullehner, 2018), as can the inhalation of particles of ammonium nitrate formed by a combination of nitrogen oxides and ammonia. OECD (2018a) provides an overview of the main externalities associated with excess nitrogen in the environment.

### Figure 2.1. Hypoxic zones in coastal waters around the world

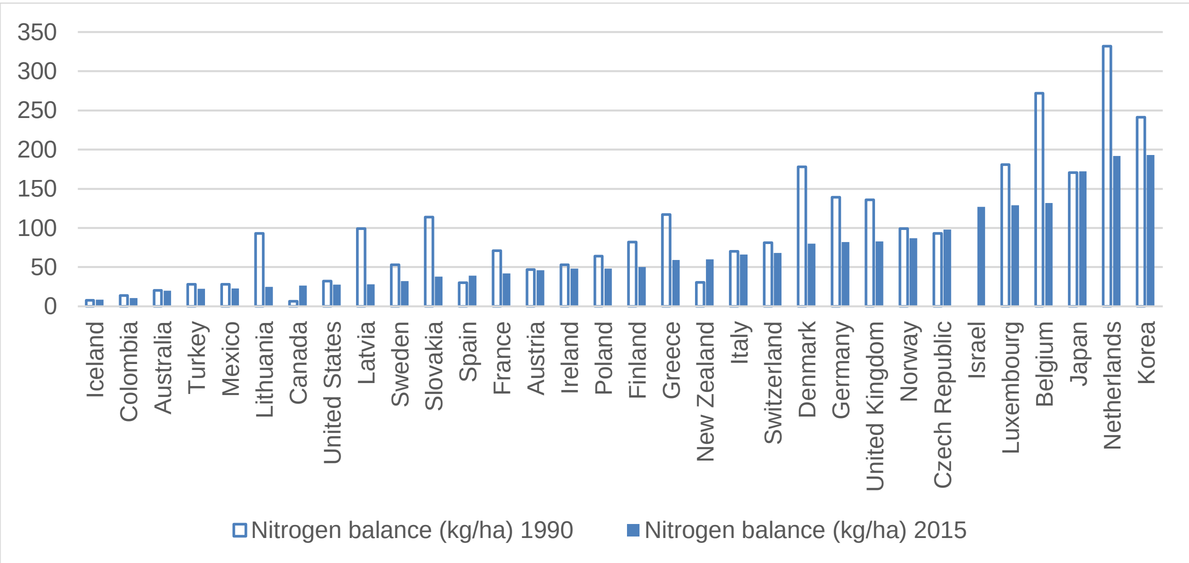



Source: Diaz and Rosenberg, 2008.

The 1990s was the decade in which several OECD countries introduced policies to manage and limit nutrient flows. The European Union's 1991 Nitrates Directive targeted specifically the agricultural sources of nitrogen losses into ground and surface waters, setting upper limits to livestock nitrogen applications in land use. The United States' Environmental Protection Agency (USEPA) in 1998 launched a national strategy based on the Clean Water Act for the setting of State water quality standards related to nutrients (USEPA, 1998 and 2020). Countries around the Baltic Sea, including Poland and Russia, in 1998 agreed an annex to the Helsinki Convention listing ten specified measures to control nutrient leaching (Thorsøe et al., 2021).

The biophysical and geochemical cycles or pathways from farmland application of nutrients to their transport and dispersion into the wider environment remains only partially and incompletely understood. The nitrogen surplus at national level is nevertheless a suitable indicator for monitoring changes in the nitrogen load resulting from efforts to manage environmental impacts of agriculture linked to the nutrient cycle. It accounts for the lost amount of nitrogen not embodied in farm products of crops etc. Data compiled by Eurostat and OECD show how, since the 1990s, the nitrogen surplus has been reduced in some OECD countries, especially within Europe, while others continue to display increasing trends (Figure 2.2). With the differences in absolute terms being huge, and despite the inherent uncertainties, these tendencies reflect how wasteful nutrient use was reduced in countries with intensive agriculture, while others on a trajectory of 'catching up' are still increasing their fertiliser use. There has been nearly a doubling of surpluses in New Zealand and Canada, whereas at the other end of the scale Belgium, Netherlands and Denmark have reduced losses with up to 50%. However, there are also countries that have continued their high levels of nutrient loss throughout the past thirty years (Korea, Japan). Trends in some of the largest countries (United states, Canada, Australia) are difficult to gauge, as national figures can mask possible nutrient 'hot-spots' at regional level.

### Figure 2.2. Agricultural nitrogen surpluses in OECD countries (2015, with changes from 1990)

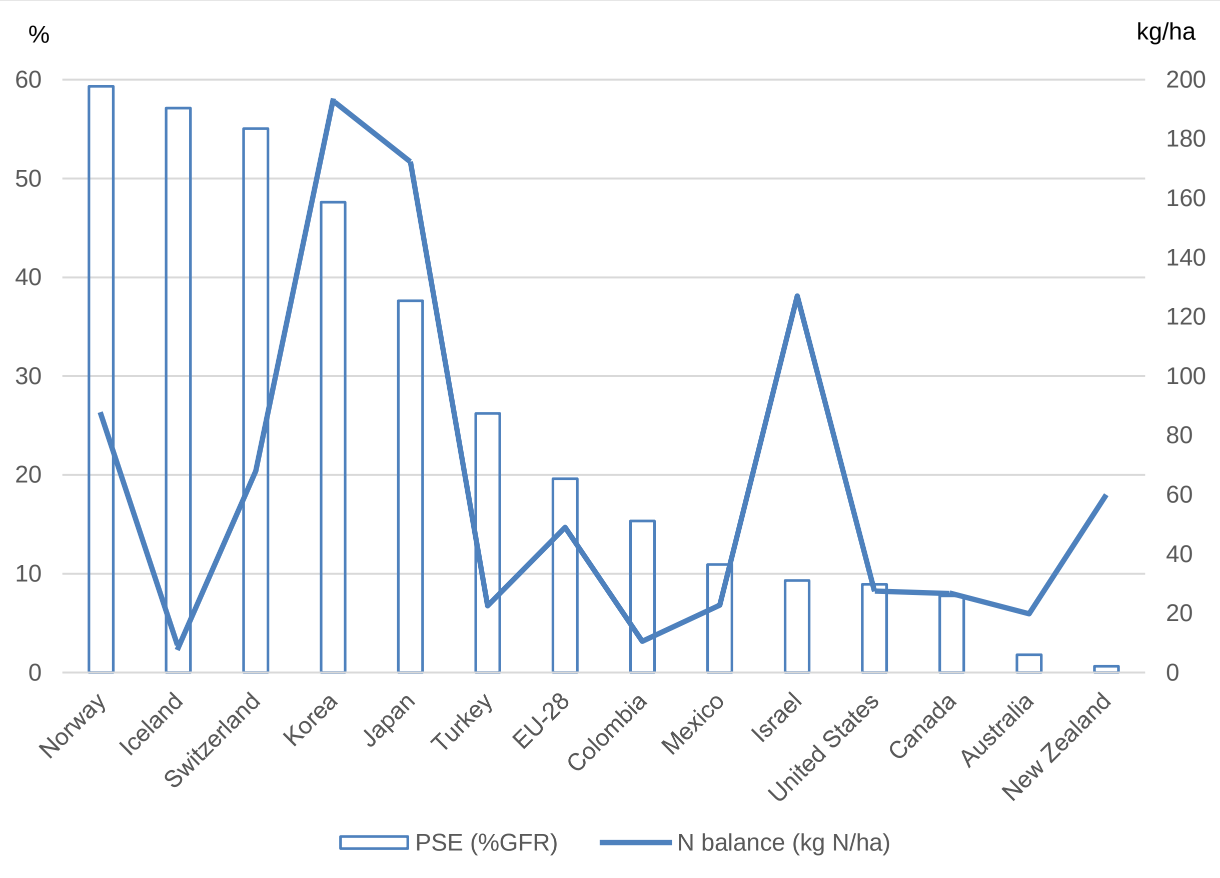



Note: The OECD soil surface nitrogen balance calculates the difference between the total quantity of nitrogen inputs entering the soil and the quantity of nitrogen outputs leaving the soil annually, based on the nitrogen cycle.
Israel: period 2012-14, data not available in 1990.
Source: OECD Stat, OECD (2019) for Israel.

With invention of the Haber-Bosch process in 1913 and its upscaling to industrial production after WW-I, nitrogen captured from the air and converted into ammonium provides hundreds of millions of tonnes of fertiliser every year for agricultural purposes. With expansion of the industry after World War II it became widely available, and its use for farming has been receiving public financial support in many parts of the world, as it has greatly leveraged food production. The most important feedstock for production of nitrogen fertilisers is natural gas, providing hydrogen for the reaction processes and energy for heating.

The level of policy transfers to agricultural producers and the surplus of agricultural nitrogen at the national level are depicted in Figure 2.3. Despite the lower level of agricultural policy support in the OECD area, New Zealand has a higher nitrogen surplus than some OECD countries with higher levels of policy support. New Zealand's nitrogen balance between 1998 and 2009 has worsened more than in any other OECD country, mainly due to the expansion and intensification of agriculture (OECD, 2017). Conversely, Iceland has the lowest nitrogen surplus in the OECD despite a substantial level of agricultural policy support. Clearly, market conditions, direct regulations as well as the specific terms of policy support play a significant role besides the climatic differences. Still, Henderson and Lankoski (2019) show that N-surpluses increase with the most distorting types of support schemes.

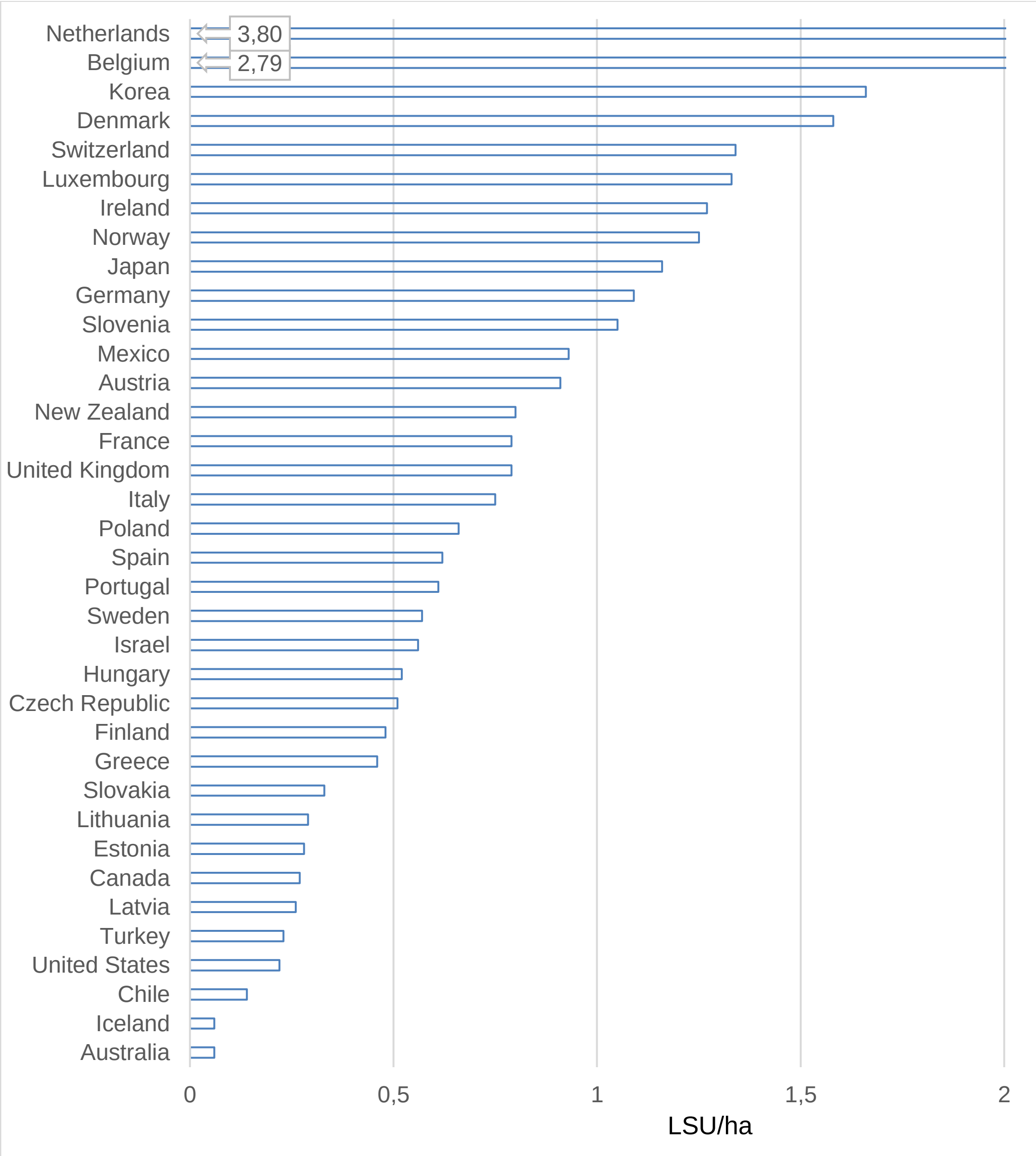
## Figure 2.3. Agricultural policy support and nitrogen surplus on the farm in OECD countries, 2015




Notes: PSE (% GFR) = policy support to agricultural producers at the farm gate level, as measured by the OECD Producer Support Estimate (PSE) as a share of Gross Farm Receipts (GFR).
N balance (kg N/ha) = soil surface nitrogen balance at the national level.
Year 2010 for the European Union. Period 2012-14 for Israel.
Source: OECD Stat, OECD (2019) for the N balance of Israel.

Another driver of nitrogen surpluses is the intensive livestock production enabled by the availability and global trade in feedstuffs, partly underpinned by mineral fertilisers (Figure 2.4) (Svanbäck et al., 2019; Huygens et al., 2020). With rising incomes and higher demand for meat, producers specialised in large-scale livestock farming have obtained competitive positions in the market. Large-scale livestock farms produce huge amounts of manure that have to be disposed of, as despite their scale these farms often do not have sufficient arable land available, a challenge that is exacerbated in densely populated countries. Farming has become specialised in many OECD countries, with dedicated crop growers acting as suppliers of domestic feedstuffs too. Such arable farms tend to prefer making use of mineral fertilisers, where nutrients are readily available for the plants, rather than to make use of manure nutrients that are more cumbersome to administrate and spread appropriately (Andersen, 2002). Losses are due both to incidental spills and to the surplus applied over crop requirements.

## Figure 2.4. Livestock densities per hectare of utilised agricultural area




Note: Reduced scale for Belgium (BEL) and the Netherlands (NLD). LSU = livestock unit.
Source: Eurostat and FAO with Eurostat methodology (Eurostat 2017).

Although aiming at securing all farmers a decent living, the EU's Common Agricultural Policy (CAP) area-support scheme tends to concentrate public support among a smaller number of farmers owing large tracts of land. Agri-environmental schemes have shown limited success in alleviating intensive modes of farming (ECA, 2020). Still, reconnecting crop and livestock through mixed farming or through opening up the organic fertiliser market -- as provided for in the EU Regulation on Fertiliser Products in the Circular Economy (EU 2019/1009) which should apply from July 16, 2022 -- will not mitigate the nutrient surpluses, unless proper account is taken of the long-run mineralisation processes of manure-N (see Chapter 3).

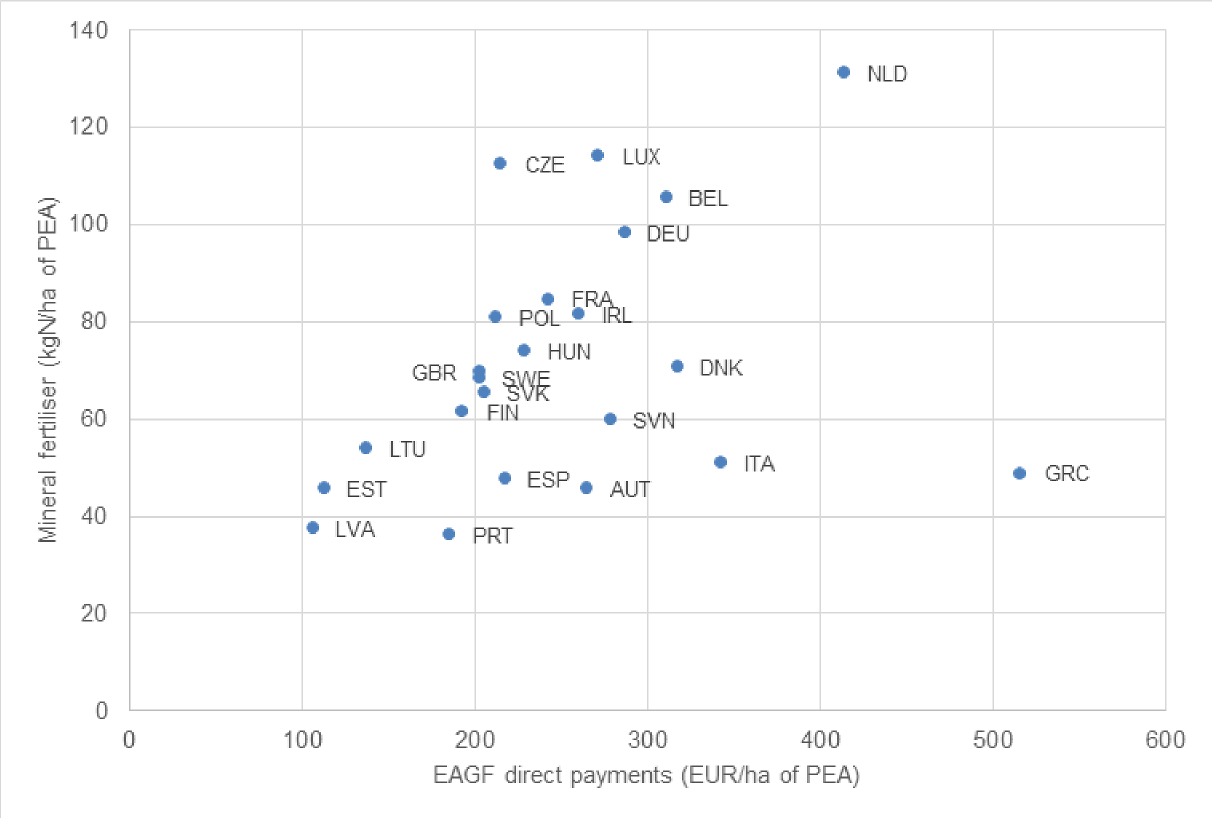
Restitution of property rights in formerly planned economies has triggered profound changes in land use practices with implications for nutrient flows. While the large-scale Soviet-style farms (kolkhozes etc.) were characterised by generous and sometimes free allocations of fertilisers, the fragmentation into smaller privately owned units, e.g. in the Baltics, lowered annual fertiliser applications. At the same time, it brought hitherto extensively farmed land and pastures under the plough, releasing run-off from soil deposits, whereby in the short term the overall relief to environment and health was diminished if not reverted. The numerous small land plots that resulted from privatisations are not sufficient to maintain a living, and in the economies in transition, many farmers cultivate their land part-time, while holding complementary occupations. They lack educational backgrounds and required skills for doing farming efficiently. In the same countries, there is often also a shortage of independent advisory services to provide guidance to farmers, who in turn tend to rely mainly on advice from the commercial suppliers of fertilisers.

Farmers in OECD countries operate in a highly competitive market, but are not all focusing equally on profit maximising their businesses. Conventional farmer professionalism is about generating high yields, with crops matching or overdoing their neighbours. Farmers that are yield maximising, rather than profit maximising, will apply nutrients more generously than optimal from an economic efficiency point of view. The marginal units of fertiliser may not secure sufficient yield increases to make up for the additional costs, and the rate of nutrient loss becomes disproportionate. These differences in farmer behaviour, nicely captured in the contrast of ‘profit versus pride’, are confirmed in large-scale surveys finding that barely 50% of farmers aim for the economic rationality normally expected, whereas about 30% declare to aim for maximum crop yield (Nielsen, 2009; Pedersen et al., 2012). A further but smaller group of farmers (about 20%) opts for less intensive modes of cultivation and production, not optimising on either yields or profits.

Fertilisers trade in a global market, whereby retail prices do not necessarily match the purchasing power of farmers that differ with productivity and levels of public financial support. Within the European Union, the highest levels of mineral fertiliser use can be observed in those countries where the CAP’s direct income support is most generous, due to the applied benchmark of historical support levels (Figure 2.5).[2] The producers of mineral fertilisers are generally exempt from taxation of their energy use - in the EU due to the ‘dual use’ exemption clause of the Energy Taxation Directive. However, emissions trading of carbon allowances may incur certain costs on fertiliser producers, despite their free allocation, as production increases need to be matched with purchases of sufficient numbers of additional allowances.

[2] Since income support is decoupled from agricultural production, its apparent association with the use of mineral fertilisers might reflect a lock-in effect of previous support schemes, even though it cannot be ruled out that the sheer provision of income support is in fact enabling farmers to purchase fertilisers in some proportion to it (wealth effect).

### Figure 2.5. Use of mineral fertilisers in the EU versus CAP income support




Note: OECD-EU countries; 2017 data.
EAGF = European Agricultural Guarantee Fund. EAGF (the "first pillar" of the CAP) provides income support to farmers as area payments decoupled from production. EAGF direct payments include the basic payment scheme (BPS) -- or the single area payment scheme (SAPS) for the Czech Republic, Estonia, Latvia, Lithuania, Hungary, Poland and Slovakia; the green payments; payments for areas with natural constraints; payments for young farmers; as well as payments under the small farmers scheme (SFS) and cotton crop specific payments (Greece). Excluding coupled support (market interventions), and national top-up payments (SAPS countries), because some of those payments are animal-based.
Mineral fertiliser = year 2014 for Belgium, Denmark, Estonia, Greece, Lithuania and Luxembourg.
PEA = Potentially Eligible Area. Total area declared by beneficiaries and potentially eligible for BPS/SAPS or SFS payment. Eligibility to the BPS/SAPS is a pre-condition to qualify for other EAGF direct payments (with the exception of coupled support).
Source: EC, 2018a and 2019a, OECD.Stat.

# 3 Abatement measures for lowering the nitrogen surplus and GHGs

An effective strategy for reducing GHGs and nutrient losses from agriculture with a win-win potential is to substitute mineral fertilisers with already available organic fertilisers (manure), while aiming to increase N use efficiency. Although this is not without behavioural and resource challenges, it can be achieved by prescribing fertiliser planning with high mandatory utilisation rates for manure-N and limitations on the use of mineral fertilisers, e.g. via taxes. With good use of manure nitrogen, there will be less mineral fertiliser costs for farmers and overall less GHG emissions (due to lower N application rates and less energy consumption for the industrial production of mineral fertilisers). Still, as manure is frequently considered a waste product presently, it seems safe to assume that substitution will have net positive implications to overall GHG emissions. The present Chapter provides a concise review of the approaches, many of which are win-win, to reduce use of mineral nitrogen fertilisers by making optimal use of organic fertilisers (manure). Although no effect for GHGs but a risk for water (eutrophication), replacing fertiliser-N with manure-N should not ignore the effect on phosphorus (P). Indeed, phosphorus is much less mobile and reactive than nitrogen, and manure recycling is bound to accumulate phosphorus in soils (Schroder, 2014).

The international literature on policy instruments for environmental regulation traditionally distinguishes four different categories of abatement measures; diffusion (‘high-stack’ policy), end-of-pipe measures, novel cleaner technology and structural measures (Simonis, 1988). However, when it comes to diffuse pollution, characterised by numerous sources and high spatial variability, diffusion is a priori the challenge, and the simplest response measure is rather containment, and then to seek means for ‘capturing’ emissions, both surpassed by cleaner technology and structural measures.

## 3.1. Containment of manure

Containment requires adequate storage facilities for manure, slurry and solid dung. It is when animal manure is spread on farmland immediately before and during the growth season of crops, that nitrogen uptake can be maximised. Since the growth season of plants is dependent on local climate, the requirements for adequate storage capacity differ with these drivers. In Northern countries up to 9-10 months of storage capacity is required, whereas countries at lower latitudes, e.g. in central Europe, will be able to manage with 6-7 months of storage capacity. Without storage capacity, nutrient utilisation rates are not likely to exceed 20% (Jensen et al., 1994).

While solid dung can be stored relatively safely with simple means (concrete floor and walls), the requirements for slurry and manure are for larger silos that can be effectively sealed off throughout the year against losses and leakages, as they are filled up. Lagoons are lacking in this respect, frequently causing contamination of groundwater reservoirs. During containment, there is a risk of methane and ammonia evaporation unless storage tanks have a sealed structure, e.g. actual roofs rather than floating covers (Hellsten, 2017).

Effective containment requires that livestock housing systems are constructed so as to enable the management of manure with a minimum of losses. Use of deep litter and loose-tie systems that are conducive for animal welfare are less efficient at containment, but good results can be obtained with slatted or drained floors, provided there is clear separation of solid and liquid manure (Nielsen et al, 2019). Methane and ammonia evaporation can be mitigated with acidification or cooling of slurry.

Containment of animal manure is to be accompanied by timing restrictions to its spreading. These restrictions can have a multitude of forms, but any post-harvest spreading of animal manure and spreading on frozen, snow-covered or water saturated soils is certainly to be avoided, to safeguard nutrient utilisation. During autumn and winter periods most regions have high precipitation rates, which tend to wash out nutrients applied during this period, as well as any inorganic residuals in excess of plant uptake from the previous growth season. Restrictions on the timing of spreading is required to underpin the containment of nutrients appropriately.

## 3.2. End-of-pipe capturing of nutrient residuals

The second category of measures are aiming directly at promoting higher uptake by plants, thus capturing the excess nutrients once they are being spread – so to say at the end of the pipe.

Cover crops that can absorb nutrient residuals during the autumn and winter period is one of the most important measures, and can be complemented by the cultivation of N-intensive crops during the growth season, e.g. beets or certain grasses. Cover crops can reduce nitrogen losses by 7-38 kg N/ha (Aronsson et al., 2016), thus allowing for win-win practices in many areas – however for the most intensively farmed countries (cf. Figure 2.2) and regions further measures will be required.

Appropriate spreading technologies are of key significance, as the conventional broad spreading of animal manure is characterised by modest utilisation rates and relatively high ammonia emissions, as well as restrictions on the possible timing of spreading, i.e. in windy conditions. While broad spreading ejects manure into the air, the use of trail hoses applies manure directly onto the ground with equipment that can be used continuously (Figure 3.1). Trail hoses require limited motor capacity, readily available on most of smaller and middle-sized farms, and the expenses can be recovered with savings on mineral fertilisers, due to 5-10% higher utilisation rates of manure nutrients being achieved (Jensen et al, 1994). One advantage of trail hoses is that spreading can be done in germinating crops, however trail hose spreading on bare soil prior to sowing is associated with relatively high ammonia emissions, which will require subsequent injection for mitigation. Alternatively, spreading can be administrated entirely with the more expensive injection technology, which can ensure even higher utilisation rates and capturing of nutrients (+5-10%), including 2-4 times less ammonia than with trail hoses

### Figure 3.1. Trail hose equipment for manure spreading

Source: SEGES, 2018.

There is an upper limit of 170 kg N/ha on the use of manure fertilisers in the EU Nitrates Directive. Where surplus manure-N is produced, it needs to be disposed of by transfer to other farmlands. However, economic calculus shows that the value of the manure nutrients (nitrogen, phosphorus, and potassium - NPK) cannot pay for the transport costs beyond distances of a few kilometres (cf. Lötjönen et al 2020). For this reason large livestock farms that concentrate manure in nutrient hot-spots are problematic (and when considering the external costs of lorry transport – about EUR 0.75 per km - the cost-effective manure spreading distances shrink further).

Buffer strips along rivers, water courses and lakes where use of fertiliser is prohibited is a preventive measure aimed at capturing nutrients, while protecting biodiversity. Buffer strips are prescribed in legislation and range from only 2 meters and up to 10 meters in different countries. Buffer strips are especially effective where the terrain is sloping, acting as a filter against excessive leaching rates, although some subsurface draining is inevitable. Buffer strips reduce the arable land available to the farmer for effective farming, but represent a reduction potential of 37-74 kg N/ha.

Mini wetlands is a novel measure aiming at capturing nutrients after drainage from the root zone, and as it is transported towards surface waters. Mini wetlands have the ability of capturing nutrients with organic carbon materials (plants, wood chips), relying on natural denitrification processes, and can be found in different varieties. To capture the drains' run-off an appropriate spatial location in the landscape is required. Mini wetlands require careful construction and management to curb the persistent GHG emissions of methane and $N_2O$, especially in the dry summer season (De Klein and van der Werf, 2014). Mini wetlands do not interfere much with regular farming activities, acting as a low-tech treatment plant requiring limited space, while offering a reduction potential corresponding to 5-20 kg N/ha. However, groundwater will not be protected.

## 3.3. Novel cleaner technology to optimise the use of nutrients

The third category of measures comprise novel cleaner technologies enabling higher utilisation rates of nutrients. Following Kemp and Rotmans (2004) these may be understood as system optimisations.

Since slurry contains much water, nutrient separation technology that results in a fibre and a liquid fraction is a means to lower the costs of transports over longer distances to arable land elsewhere. Several relatively simple, mechanical technologies are available (screw presses, decanter centrifuges and combinations of these) while the use of chemical additives is an option too (Tybirk et al., 2013). Separation technologies do not reduce the amount of nutrients, but by separating out a more nutrient rich fibre fraction that is easier to store, manage and transport they increase the range of manure use, and thus of the possible mineral fertiliser substitution. These technologies can enable a distributed nutrient management system (soluble N and K in the liquid fraction; slowly available organic N and most of the P in the solid fraction). Processed manure characterised by a high plant available N ratio may have similar agronomic efficiency as mineral fertiliser-N (Huygens et al., 2020). However, the use of separation technology requires significant investments, proving attractive mainly where land prices have soared, as it will allow the farmer to increase his livestock production beyond existing regulatory requirements for land. It has also become attractive in relation to supplying biogas plants with manure (see below).

Precision agriculture based on advanced digital tools has been advocated as a further means to reduce fertiliser use. With the spatial mapping of the variability in yields across fields as relating to a number of key variables in relation to terrain, organic matter content, moisture levels etc. it enables the farmer to adjust the administration of nutrients in a targeted way. While it has shown potential, it might be too optimistic about the capacity of the majority of farmers to transform this wealth of data into their day-to-day management, which requires proficient advisory support. Pilot-experiments suggest that it applies to use of mineral fertilisers easily, whereas application of organic fertilisers is associated with several other practical constraints that limit the opportunities for precision (SEGES, 2018).

Use and processing of slurry manure for the production of biogas helps to increase nutrient availability as the total ammonia nitrogen (TAN) digestate that is returned as fertiliser to arable land is 5-10% higher than in untreated manure (thus injection to minimise ammonia losses is useful) (Huygens et al., 2020). However, due to low C availability in slurry it is necessary to complement with other feedstock, e.g. from industry or households. The produced methane can be used for combined production of power and heat in a conventional unit, but it can also be upgraded to gas quality for distribution in the gas grid, where it can substitute fossil fuel based gas. The upgrading process emits $CO_2$, which however can be captured for industrial uses. In the future it might be possible to store wind energy as methane by using (excess) electricity to feed a process where the biogas $CO_2$-emissions are converted with hydrogen to methane. One caveat with biogas production are the concurrent leakages of methane, which it is difficult to eliminate. Many of the small-scale biogas plants with only a part-time operator have been found to have high leakage rates (about 5%), whereas best practice aimed for by the large plants is to reduce leakage rates to 1%. Thus, climate neutral biogas production would seem to require large-scale, professional operators rather than farm-level biogas plants, as also preferred by financial investors.

## 3.4. Structural measures to reduce nutrient volumes

The fourth category of measures is structural adjustments of agriculture.

One evident measure is to limit the number of animals raised per hectare of utilised agricultural area, in order to limit the amount of animal manure to match the cropland available with reasonable transport costs (Brady et al., 2021). The EU's Nitrates Directive comes close to doing so, by prescribing a maximum amount of livestock manure that can be applied each year of 170 kg N/ha. The Helsinki Convention on the Protection of the Marine Environment of the Baltic Sea Area goes one step further, by prescribing that countries establish an animal density, reflecting a balance between the number of animals and the amount of land available for spreading manure, to ensure that manure is not produced in excess in comparison to the amount of arable land (Thorsøe et al., 2021). With one livestock unit corresponding to 100 kg N per year, the resulting standard harmony requirement is a maximum of 1.7 animal units per hectare, though with different national excretion factors. There are possible derogations when farmland is available for spreading manure through contracting with neighbours. Some countries have also obtained higher limits (up to 2.3 livestock units for cattle) under the exemption procedures of the Nitrates Directive. Nevertheless, a cap on the number of animals, with prescription of an animal density per hectare, seems to be a simple and enforceable regulatory requirement.

Increasing the share of organic farming constitutes a further structural measure to increase nutrient cycling and reduce the applications of mineral fertilisers. This is because organic farmers according to their codex will not apply mineral fertilisers, but only organic fertilisers, thus ensuring that these fertilisers are used effectively and are not considered as waste. Still, the productivity of organic farming is lower and it entails some nutrient losses as well, depending on the utilisation rate aimed for with regard to organic manure. A recent analysis finds a reduction potential with organic farming of 6-9 kg N/ha as compared to conventional farming with catch crops (Olesen et al., 2020).

Other alternative types of farming will qualify as measures relevant to lower the use of mineral fertilisers. One example is conservation agriculture, which aims at limiting soil tillage. Conservation agriculture aims to maintain permanent soil cover with crop residues and live mulches. It enhances biodiversity and natural biological processes contributing to nutrient use efficiency, although application of mineral fertilisers is not ruled out, but they must be applied optimally in a way that is compatible with biological processes.

The category of structural adjustments comprises furthermore measures that reduce the share of agricultural land, converting it to forestry, nature areas or undisturbed peatlands that have modest leaching rates of nitrogen. With demand unchanged, this will cause displacement of production to other locations, which however may be less intensively farmed.

While the above structural measures would tend to downscale agricultural intensity, structural measures may also be understood as innovation at system level, cf. Kemp and Rotmans, 2004.

Extending from biogas production based on manure it would be possible to establish a dedicated bio-refinery, where harvested domestic grasses are used for extraction of protein, while the residuals of fibre and liquid provide additional feedstock for the biogas plant. The grass protein could replace soy protein demand for cultivation in tropical areas, thus contributing positive land use and GHG emission implications at the global level, while the biogas digestate would replace mineral fertilisers. Growing grasses rather than crops would reduce nitrogen surplus as well, despite increased use of fertilisers (Manevski et al., 2018). This approach would imply changes in the use of arable land, replacing conventional crops with grasslands. Recent experience from pilot plants suggests that bio-refineries will require substantial public financial support to become viable (Martinsen and Andersen, 2020).

A fundamentally different conceptual vision involves combining the manure-based fibre fraction from biogas with straw and other crop residues to produce bio-oil and syngas through well-known pyrolysis processes. Thus, GHGs will be transformed into useful biofuels before being emitted, which allows for substitution of conventional fossil fuels. With the pyrolysis process about half the GHG emissions could be captured in a waste stream of bio-char, a stable compound which can be used to improve soil quality, thus reducing leaching, while acting as a sink for carbon (Collins, 2019). One option would be to convert the resulting fuels to methanol through reaction with hydrogen, which could be generated with GHG-neutral wind power, as envisioned with the SkyClean technology, which aims at a further upgrade to jet-fuel quality (Leonhard, 2019). According to the inventor, this technology could from the manure-based biogas fibre fraction supply jet-fuel which is carbon-negative and at a price of about USD 50/barrel of oil equivalent. However, the fate of N compounds during the process of pyrolysis warrants further research.

# 4 Carbon footprints of mineral fertilisers

The above measures do not target reductions in demand for agricultural products, which is regarded as exogenous.

Mineral fertilisers are associated with emissions of $CO_2$ and $N_2O$ in the production phase as well as in the cultivation phase (including transport). Previous estimates by Fertiliser Europe suggested a carbon footprint of about 9-10 kg $CO_2$-eq per kg of nitrogen when considering the entire lifecycle (Brentrup and Palliere, 2008). Table 4.1 shows updated values for GHG emissions related to the production phase, thus not including $N_2O$ from the cultivation phase.

Implementation of the emissions trading system for carbon emissions in Europe (EU ETS) has triggered improvements, so that via production process changes (higher energy efficiency and $N_2O$ abatement equipment) by 2014 the carbon footprint is reduced with about 1.4 kg $CO_2$-eq per kg N (cf. Brentrup et al., 2018). This represents a production phase efficiency improvement of about 20%, and is additional to the more substantial improvements in earlier years, whereby the total production related improvement in Europe since 1990 is estimated to be in the range of 40-60% (37-43% for urea ammonium nitrate and 60% for ammonium nitrate). According to global figures of fertiliser production GHGs, most other world regions have not implemented comparable adjustments to their production processes, and emissions have been maintained at levels that now count as historical in Europe.

Table 4.1 shows for 10 world regions the estimated carbon footprint values of mineral fertiliser production. China has, based on lifecycle analysis, a carbon footprint about 3 times as high as Europe due to the use of coal for energy supply to production, while other world regions, including North America, have a carbon footprint that is twice as high.

The cultivation phase emissions of $N_2O$ are unaffected by the above innovations, but the fertiliser industry has been promoting stabilised fertilisers as a novel technology to reduce GHG emissions in the cultivation phase. Stabilised mineral fertilisers are provided with a coating that temporarily inhibit nitrogen transformation from which $N_2O$ may be released. Table 4.2 provides data for the GHG emissions associated with mineral fertiliser production and use in Germany. The data reveals that reductions of up to 13% could be achieved with stabilised fertilisers, however more needs to be understood about the wider environmental effects. The study by Hasler et al (2017) additionally considered the combination of fertilisers with irrigation (fertigation) as well as secondary raw material fertilisers. It confirms that $CO_2$ emissions are as important as $N_2O$ for a lifecycle assessment of the GHGs of mineral fertilisers, as for instance ammonium nitrate (AN) with a $N_2O$ emission of 596 kg $CO_2$eq per functional unit in cultivation also involves $CO_2$ emissions of 452 kg $CO_2$ + 105 kg $CO_2$ during production and cultivation – besides a production phase $N_2O$ emission of 150 kg $CO_2$eq and 10 kg $CO_2$ in transport. The functional unit refers to 125 kg N/ha. Fertilisers made from secondary raw materials (feather, bones, leguminous crops) have similar carbon footprints as mineral fertilisers, but can make good use of waste products in a way that can also help improve soil quality.

## Table 4.1. Reference carbon footprint values for main mineral fertiliser products (kg $CO_2$eq/kg N)

| Fertiliser product[1] | Nutrient content (%) | Region[2] | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | EU-27 | EECCA | Africa | Middle East | North America | Latin America | China | South Asia | South-East Asia | Oceania |
| Ammonium nitrate (granulated) | 33.5N | 1.14 | 2.42 | 2.10 | 2.44 | 2.28 | 2.17 | 3.50 | 2.32 | 2.39 | 2.09 |
| Ammonium nitrate (prilled) | 33.5N | 1.11 | 2.38 | 2.06 | 2.40 | 2.25 | 2.13 | 3.44 | 2.27 | 2.34 | 2.05 |
| Ammonium sulphate | 21N, 24S | 0.56 | 0.68 | 0.62 | 0.59 | 0.61 | 0.66 | 1.12 | 0.81 | 0.65 | 0.56 |
| Ammonium sulphate nitrate | 26N, 14S | 0.80 | 1.39 | 1.22 | 1.33 | 1.29 | 1.27 | 2.08 | 1.42 | 1.35 | 1.18 |
| Anhydrous ammonia[3] | 82N | 2.30 | 2.67 | 2.38 | 2.19 | 2.49 | 2.56 | 4.20 | 2.92 | 2.38 | 2.05 |
| Calcium ammonium nitrate | 27N | 0.95 | 1.98 | 1.72 | 2.00 | 1.87 | 1.78 | 2.86 | 1.90 | 1.96 | 1.72 |
| Calcium nitrate | 15.5N | 0.64 | 1.66 | 1.44 | 1.75 | 1.58 | 1.47 | 2.34 | 1.53 | 1.68 | 1.48 |
| NPK (mixed-acid route) | 15N, $15P_2O_5$, $15K_2O$ | 0.62 | 1.13 | 1.00 | 1.13 | 1.06 | 1.03 | 1.61 | 1.13 | 1.13 | 1.01 |
| NPK (nitrophosphate route) | 15N, $15P_2O_5$, $15K_2O$ | 0.71 | 1.22 | 1.09 | 1.22 | 1.15 | 1.11 | 1.71 | 1.24 | 1.22 | 1.10 |
| Diammonium phosphate | 18N, $46P_2O_5$ | 0.63 | 0.75 | 0.70 | 0.68 | 0.67 | 0.73 | 1.15 | 0.89 | 0.74 | 0.66 |
| Monoammonium phosphate | 11N, $52P_2O_5$ | 0.44 | 0.53 | 0.51 | 0.51 | 0.46 | 0.52 | 0.81 | 0.66 | 0.55 | 0.51 |
| Super phosphate | $18P_2O_5$, 12S | 0.08 | 0.09 | 0.10 | 0.11 | 0.08 | 0.09 | 0.13 | 0.13 | 0.11 | 0.11 |
| Triple super phosphate | $48P_2O_5$ | 0.18 | 0.21 | 0.22 | 0.24 | 0.18 | 0.21 | 0.27 | 0.28 | 0.25 | 0.25 |
| Urea[4] | 46N | 0.88 | 1.10 | 0.93 | 0.81 | 1.01 | 1.01 | 1.99 | 1.27 | 0.93 | 0.75 |
| Liquid urea ammonium nitrate solution (UAN)[4] | 30N | 0.78 | 1.43 | 1.23 | 1.34 | 1.33 | 1.29 | 2.20 | 1.44 | 1.36 | 1.17 |
| Limestone[5] | $55CaCO_3$ | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 |
| Potassium chloride[5] | $60K_2O$ | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 |
| Potassium sulphate[5] | $50K_2O$, 18S | 0.12 | 0.12 | 0.12 | 0.12 | 0.12 | 0.12 | 0.12 | 0.12 | 0.12 | 0.12 |

Notes: 1. Excluding cultivation and transport; reference year: 2014.
2. EECCA = Eastern Europe, Caucasus and Central Asia (Belarus, Russia, Turkmenistan, Ukraine, Uzbekistan); Africa (Algeria, Egypt, Nigeria, South Africa); Middle East (Iran, Kuwait, Oman, Qatar, Saudi Arabia, Turkey, United Arab Emirates); North America (United States, Canada); Latin America (Argentina, Brazil, Mexico, Trinidad & Tobago, Venezuela); South Asia (India, Pakistan); Southeast Asia (Indonesia, Malaysia, Vietnam); Oceania (Australia, New Zealand).
3. Ammonia as intermediate product from Haber-Bosch process is used as a proxy for anhydrous ammonia i.e. no further processing of ammonia considered.
4. Excluding emissions from urea hydrolysis: 0.73 kg $CO_2$/kg urea or 0.24 kg $CO_2$/kg UAN will be released after application of the product.
5. For these products global average default emission factors are assumed.
Source: Brentrup et al., 2018.

## Table 4.2. GHG emissions during the production, transportation and application of mineral fertilisers (Germany)

| Fertiliser product[1] | N content (%) | Production | | | |
|---|---|---|---|---|---|
| | | Kg $CO_2$ | Kg $N_2O$ | Kg $CO_2$ | Kg $N_2O$ as $CO_2$eq |
| | | /kg fertiliser | | /FU[2] | |
| Ammonium nitrate | 35 | 1.26 | 0.00141 | 452 | 150 |
| Ammonium nitrate fertigation | 35 | 1.26 | 0.00141 | 452 | 150 |
| Calcium ammonium nitrate | 27 | 0.98 | 0.00116 | 456 | 160 |
| Calcium nitrate + fertigation | | 0.98 | 0.00341 | 456 | 464 |
| Urea ammonium nitrate (UAN) | 32 | 1.09 | 0.00070 | 426 | 82 |
| Urea | 46 | 1.42 | 0.00004 | 386 | 3 |
| Urea + urease inhibitor | 46 | 1.42 | 0.00004 | 386 | 3 |
| Urea + urease inhibitor + nitrification inhibitor | 46 | 1.42 | 0.00004 | 386 | 3 |
| Urea + fertigation | 46 | 1.42 | 0.00004 | 386 | 3 |
| Ammonium sulphate nitrate | 26 | 0.93 | 0.0561 | 453 | 465 |
| Ammonium sulphate nitrate + nitrification inhibitor | 26 | 0.93 | 0.0561 | 453 | 465 |
| Feather meals | 0.13 | 0.14 | 0.00140 | 255 | 734 |
| Meat and bone meals | 0.10 | 0.68 | 0.00300 | 1 088 | 1 445 |
| Leguminous crop meals | 0.05 | 0.14 | 0.00001 | 850 | 7 |

| Fertiliser product[1] | Transportation | | Cultivation | | | |
|---|---|---|---|---|---|---|
| | Kg $CO_2$ | | Kg $CO_2$ | | Kg $N_2O$ | Kg $N_2O$ as $CO_2$eq |
| | /tkm | /FU[2] | /kg urea-N or nitrate N | /FU[2] | /kg fertiliser | /FU[2] |
| Ammonium nitrate | 0.037 | 10.5 | 0.84 | 105 | 0.0054 | 596 |
| Ammonium nitrate + fertigation | 0.037 | 10.5 | 0.84 | 105 | 0.0050 | 534 |
| Calcium ammonium nitrate | 0.049 | 13.8 | 0.84 | 105 | 0.0042 | 596 |
| Calcium nitrate + fertigation | 0.048 | 12.9 | 0.84 | 105 | 0.0028 | 379 |
| Urea ammonium nitrate (UAN) | 0.041 | 11.6 | 0.94 | 333 | 0.0050 | 581 |
| Urea | 0.027 | 7.90 | 1.60 | 446 | 0.0071 | 596 |
| Urea + urease inhibitor | 0.027 | 7.90 | 1.60 | 446 | 0.0068 | 551 |
| Urea + urease inhibitor + nitrification inhibitor | 0.027 | 7.90 | 1.60 | 446 | 0.0045 | 364 |
| Urea + fertigation | 0.027 | 7.90 | 1.60 | 446 | 0.0045 | 370 |
| Ammonium sulphate nitrate | 0.034 | 10.7 | 0.84 | 105 | 0.0041 | 461 |
| Ammonium sulphate nitrate + nitrification inhibitor | 0.034 | 10.7 | 0.84 | 105 | 0.0027 | 367 |
| Feather meals | 0.015 | 25.8 | 0.84 | 105 | 0.0011 | 596 |
| Meat and bone meals | 0.015 | 24.5 | 0.84 | 105 | 0.0012 | 596 |
| Leguminous crop meals | 0.008 | 53.8 | 0.84 | 105 | 0.0003 | 596 |

Notes: 1. In one cultivation period; including fertilisers applied via fertigation and fertilisers made from secondary raw materials.
2. FU = functional unit in cultivation, representing an application of 125 kg N per hectare.
Source: Hasler et al., 2017.

With regard to making use of manure, as discussed in Chapter 3, the key challenge is to take account of the amounts of nitrogen available to plants with slurry and solid manure respectively and according to the livestock origin. Table 4.3 provides data for the N-contents of manure and the standard utilisation rate requirements for fertiliser planning in Denmark. The legally required utilisation rates of total-N exceed the shares of inorganic (ammonia) nitrogen available in the initial growth year, whereby farmers must factor in a share of the subsequent mineralisation from previous years' applications.[3] Note that requirements apply to solid manure too.

### Table 4.3. Contents of inorganic and organic N in manure and legally required utilisation rates in Denmark

| Type of manure | Ammonia-N (%) | Organic-N (%) | Total-N (%) | Utilisation demand of total-N (%) |
|---|---|---|---|---|
| Pig slurry | 75 | 25 | 100 | 80 |
| Cattle & poultry slurry | 58 | 42 | 100 | 75-80 |
| Solid manure, pigs | 45 | 55 | 100 | 55 |
| Solid manure, cattle | 30 | 70 | 100 | 55 |

Source: Circular 1166/2020 of Ministry of Environment and Food (Denmark).

The biophysical mechanisms in soil and plant uptake of nitrogen are not completely understood (for a good summary of knowledge see OECD, 2018a), however based on field experiments with nitrogen it has been possible to establish a basic understanding of the mineralisation processes. Table 4.4 shows for a 10-year period estimated rates of mineralisation and the accumulated after-effect of manure nitrogen. N-residuals gradually mineralise into inorganic N available for plants. About half of the N-residuals (45%) will remain as organic compounds after 10 years, or may have dissipated into other biophysical processes, however about 55% will have mineralised and become available as inorganic N, although the specific mineralisation rates will depend on temperature and climate.

### Table 4.4. Mineralisation of inorganic N after manure application in the Danish climate

| Year of after-effect | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
|---|---|---|---|---|---|---|---|---|---|---|
| Mineralisation rate % | 20 | 12.0 | 8.0 | 6.0 | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 | 5.0 |
| Mineralisation rate of initial % | 20 | 9.6 | 5.6 | 3.9 | 3.0 | 2.9 | 2.7 | 2.6 | 2.5 | 2.4 |
| Cumulative mineralisation % | 20 | 29.6 | 35.2 | 39.1 | 42.2 | 45.1 | 47.8 | 50.4 | 52.9 | 55.2 |
| Organic N-residuals % | 80 | 70.4 | 64.8 | 60.9 | 57.8 | 54.9 | 52.2 | 49.6 | 47.1 | 44.8 |

Source: Based on annex D in Petersen and Sørensen, 2008.

These are conservative estimates (see Sørensen et al, 2017) but despite Denmark's northern latitude nevertheless exceed the utilisation rates defined as part of good agricultural practices in other EU Member States (see Annex A) and which are optional to farmers.

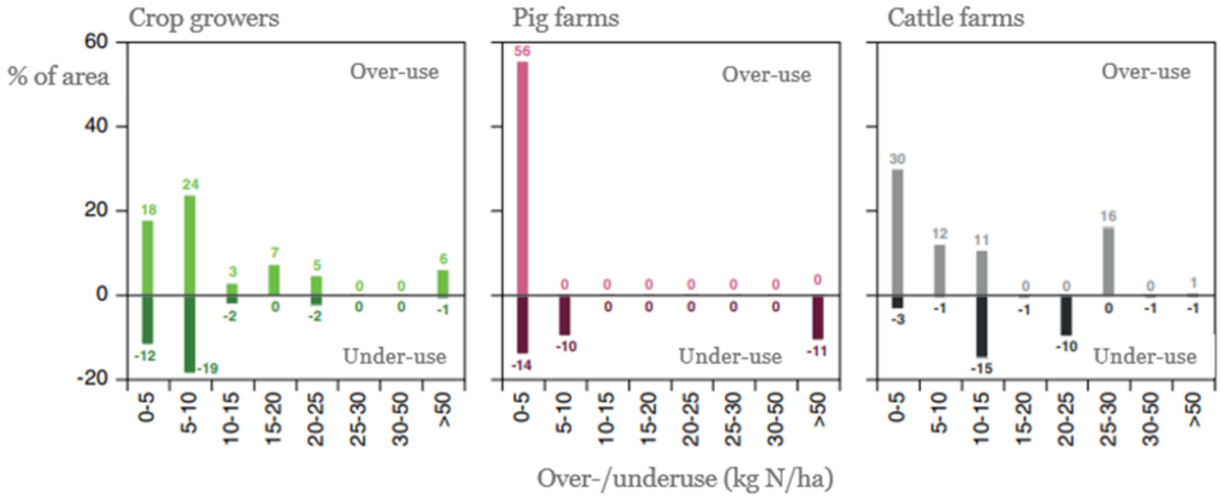
Croplands that have been fertilised for many years with manure, without due account being taken of its fertiliser value, will have accumulated a large deposit of organic nitrogen in soils. These deposits will leave a fingerprint on the annual nitrogen leaching rates, that will be reflecting as well farm practices in the current year as the legacy of previous years, as N-residuals mineralise into inorganic N.

With the high shares of residual-N in solid manure (55-70%), there is a key challenge in substitution of mineral fertiliser with manure, so that not only the fertiliser value of the initial year, but also that of the subsequent years is taken into account. Also slurry has substantial shares of organic N-residuals that should not be neglected (25-42%), especially for cattle. However, cattle farmers are more frequent violators of respecting the balance between crop needs and fertiliser application, as confirmed by monitoring; Figure 4.1 shows that 27% (16% + 11%) of cattle farm areas are over-fertilised. This might reflect the

[3] Planning and bookkeeping of fertiliser applications is mandatory and to be entered into an online reporting system to the authorities. Applications of mineral fertilisers have been reduced by about 50% since 1990 due to these requirements.

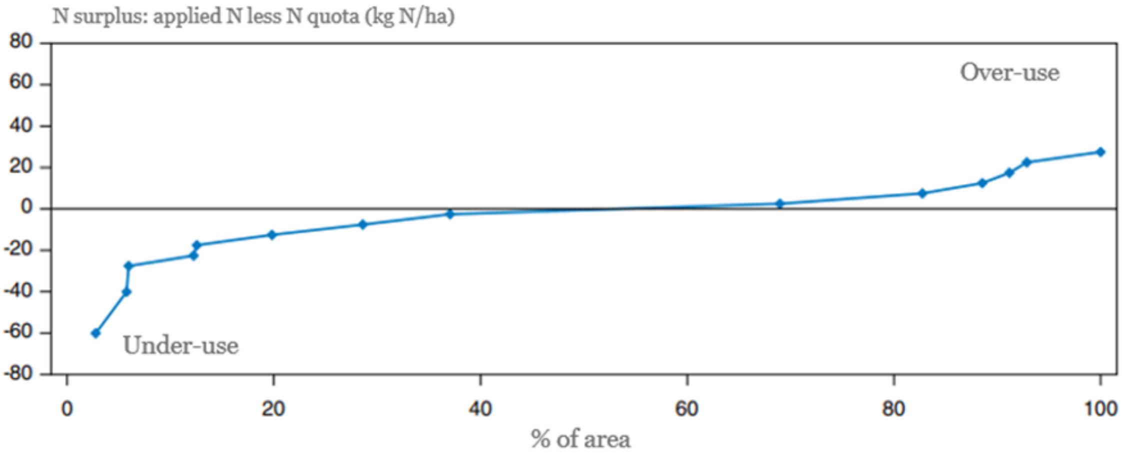
difficulties of factoring in the long-run mineralisation processes. In total 15% of farmland, including also a share cultivated by non-livestock holding crop growers (Figure 4.2), is found to receive more than 10 kg N/ha in excess of the crop quota for nitrogen use, according to regulatory requirements based on the prescribed utilisation rates. The nitrogen use monitoring results refer to field blocks at the sub-farm level, whereby underuse may relate to distant fields for which manure transports are too costly and burdensome. The non-compliance rates might be expected to provide a conservative estimate among the subset of farmers subject to predictable, annual monitoring.

### Figure 4.1. Over- and underuse of nitrogen relative to the N-quota in Denmark according to the type of livestock




Source: Grant et al., 2011.

### Figure 4.2. Land areas with over- and underuse of nitrogen relative to the N-quota in Denmark




Note: Among farms subject to Denmark's National Monitoring and Assessment Programme for the Aquatic and Terrestrial Environment (NOVANA).
Source: Grant et al., 2011.

# 5 Policy instruments for sustainable nutrient management

OECD countries with medium and low densities of livestock tend to have housing systems that mainly produce solid manure, whereas countries with high livestock densities tend to produce slurry and/or liquid manure. As a paradox there might still be huge nitrogen losses, if farmers are unaware of or neglect the long-run value of manure, despite their modest livestock densities.

The policy instrument literature distinguishes between information, direct environmental regulation and economic policy instruments, including pricing instruments (taxes, emissions trading systems) and financing instruments (public financial support, payments for ecosystem services) (OECD, 2012). The predominant way to address nutrient is to use informational and regulatory approaches (command-and-control). Economic policy instruments in terms of public financial support as well as taxes are used occasionally. The former has been provided in several different ways, e.g. direct support as well as tax credits to farmers. The present Chapter will briefly review some of the main approaches observed in some OECD countries.

The EU's Nitrates Directive of 1991 requires that Member States establish codes of 'good agricultural practices' (GAP) to be implemented by farmers. These codes must identify the periods during which application of fertiliser is inappropriate and the conditions for applications to sloping grounds, near water courses or during periods of flooded or frozen ground. Moreover, the codes must specify the capacity requirements for storage of manure and the procedures for spreading. Codes may also (optionally) prescribe the use of cover crops, crop rotations, fertiliser plans, nutrient book-keeping and other preventive measures. For large-scale livestock facilities permits are required under the Industrial Emissions Directive (IED) to ensure that Best Available Technology is implemented.

Based on identification of waters vulnerable to leaching, Member States are moreover required to designate Nitrate Vulnerable Zones (NVZ) where action programmes must be developed, whereby a regulatory approach will apply. Under such NVZ action programmes the above mentioned GAP codes are mandatory, if not superseded by any of the additional directive requirements, which include a cap of 170 kg N/ha for the spreading of manure. Furthermore, they make it mandatory to have storage capacity sufficient to match the longest period during which spreading is prohibited, and there is a balancing requirement stating that fertiliser use should not exceed nitrogen requirements of crops, while taking into account soil deposits and net mineralisation of nitrogen. Member States may choose to NVZ designate their whole territory.

The entire national territory has been NVZ-designated in ten countries; Austria, Denmark, Finland, Germany, Ireland, Lithuania, Malta, Romania, Slovenia and very recently Poland (EC, 2018b). Notwithstanding this approach, the nitrogen surpluses of these countries suggest that they are far from compliance with the fertilisation planning requirement of the Nitrates Directive. Germany's long-time policy legalising a surplus of up to 60 kg N/ha was recently overruled by the Court of Justice of the EU, which has triggered a profound revision of Germany's National Fertiliser Ordinance – NFO (CJEU, 2018).

In the context of the Helsinki Convention on the Protection of the Marine Environment of the Baltic Sea Area somewhat more stringent measures have been agreed by the nine littoral states, of which all apart from Russia are now EU Member States.[4] Agreed in 1998, annex III part 2 specifies ten measures addressing nutrients, to be transposed into national law. They go beyond the Nitrates Directive, e.g. with a balancing requirement concerning the number of animals relative to land (animal density) and a minimum of six months requirement for manure storage. While repeating the manure application limit of 170 kg N/ha, the balancing requirement of fertiliser use relative to crop needs is to be specified in national guidelines. Organic manure is to be incorporated after spreading, cover crops are required and buffer/protection zones to be established. Item 10 has requirements to limit ammonia emissions from livestock with national programmes. From 2018 large livestock installations are to do nutrient bookkeeping (Thorsøe et al., 2021).

In the United States, non-point source pollution from agriculture is largely outside the scope of the Clean Water Act requirements, except that large livestock units (Concentrated Animal Feeding Operations, CAFOs) must obtain pollution permits as point sources and implement comprehensive nutrient management plans (NMP). Since at the federal level nutrients are addressed mainly through voluntary programmes, the USEPA and the states have initiated a partnership to reduce non-point source pollution (Figure 5.1), which makes approved state programmes eligible for USEPA financial support (OECD, 2006). The National Water Quality Initiative, a joint initiative of USEPA, the United States Department of Agriculture (USDA) and state water quality agencies, has selected priority watersheds in all states with targeted on-farm investments to limit non-point source pollution, including from nitrogen, but covering only a few percent of lands. With regard to the NMP's for large livestock units, requirements are of the command-and-control type, prescribing a balancing requirement between crop nutrient needs and total fertiliser availability, based on the development of a detailed nutrient budget. In determining the available nutrients from previous years, standard mineralisation rates are prescribed. A nitrogen index classifies farmlands and their leaching risk, with implications for NMP modifications where risks are considered high.

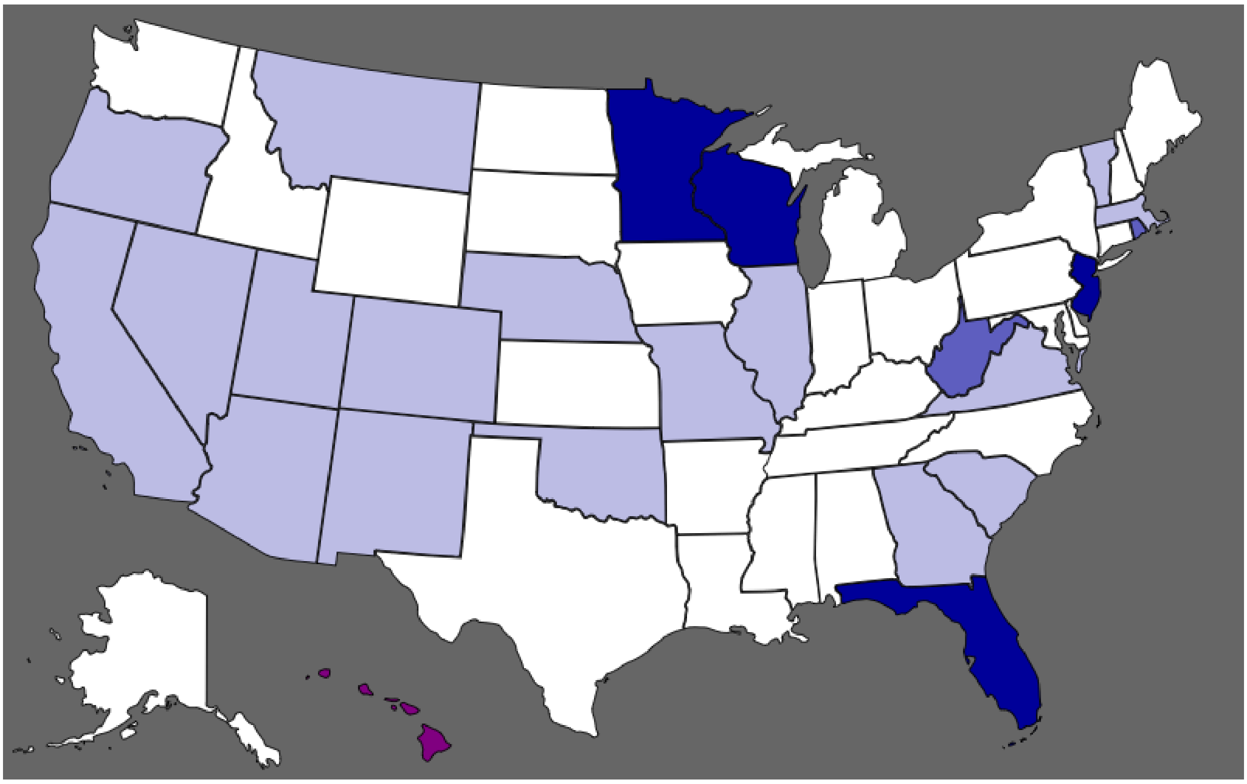
Some USDA general programmes have nutrient implications; this is the case with the Working Lands programmes -- including the Environmental Quality Incentives Program (EQIP) and the Conservation Stewardship Program (CSP) -- and the Land Retirement programmes -- including the Conservation Reserve Program (CRP). The former provide financial and technical assistance for cover crops, buffer zones and precision technology, while the latter offer payments for conservation covers on sensitive lands (Sud, 2020). Some US states are making tax credits available to farmers. Oklahoma provides poultry farmers with incentives to sell manure as fertilisers rather than disposing of it near waterways, allowing buyers of manure to claim a tax credit of USD 5 per tonne. Oregon provides dairy farmers a tax credit for supplying manure to produce biofuel.

Canada pursues a bottom-up strategy for reducing nutrients, relying on self-administered risk-assessment tools in terms of Environmental Farm Plans, that assist farmers in developing customized action plans to address nutrient leaching risks on their farms. Financial support is provided from the federal level to complement resources allocated by Provincial governments under their Environmental Stewardship Incentive programmes to accelerate the diffusion of Environmental Farm Plans across Canada, and emphasizing the role of farmers in optimising the N application rate and in delivering higher nutrient use efficiency (OECD, 2015b; Smith et al., 2020).

New Zealand provides financial compensations to farmers for permanent reductions in nitrogen use below a defined cap in vulnerable lake districts (Lake Taupo since 2011, Rororua Lakes since 2017). The compensations are financed through a taxpayer-funded Trust, which buys excess nitrogen allowances to permanently withdraw them from the market (Duhon, McDonald and Kerr, 2015). The Trust Fund also issues credits for carbon sequestration associated with the conversion of agricultural land to forests, with the government mandating these credits for sale into the national emissions trading system (OECD, 2015a; 2018a).

[4] Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Sweden.

### Figure 5.1. US states with nutrient water quality criteria for nitrogen and phosphorus.



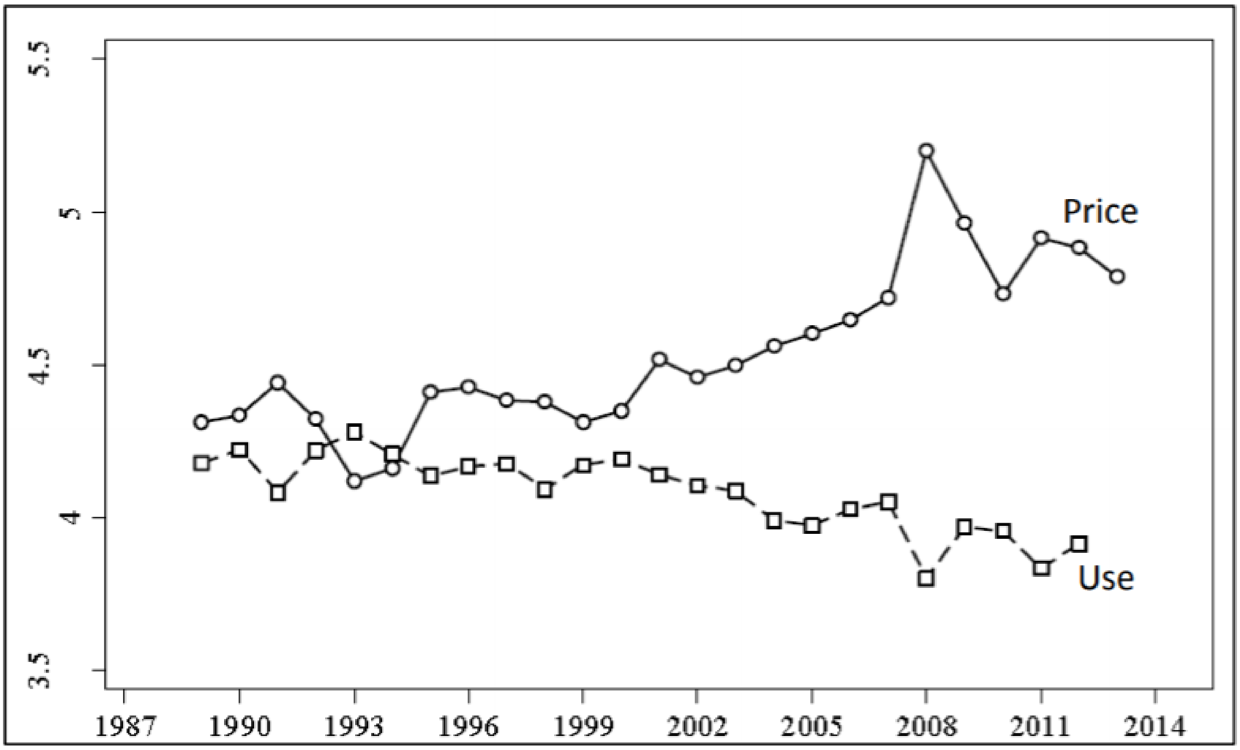
Note: Red = Level 5: complete set of N and P criteria for all water types; Dark blue = Level 4: two or more water types with N and/or P criteria; Blue = Level 3: one water type with N and/or P criteria; Light blue = Level 2: some waters with N and/or P criteria; White = Level 1: No N and/or P criteria. The three "water types" refer to lakes/reservoirs, rivers/streams, and estuaries.
Source: USEPA, 2020.

In Australia, the reduction of surplus nutrients entering the Great Barrier Reef (GBR) is a key goal in the Reef 2050 Water Quality Improvement Plan. Information measures (monitoring and modelling at the paddock and watershed scale) identified that the cultivation of sugarcane was the main pathway for nitrogen to the GBR. A combination of regulations (compulsory farm nitrogen budget for sugarcane producers from December 2021) and financial incentives up to AUD 500 000 (EUR 309 000) per farm aim to reduce excessive fertiliser use and to manage the risks of soil erosion in the GBR catchments. Financial incentives are awarded through competitive bidding (reverse auctions) to allocate funds from a Reef Trust, which is funded by taxpayers and developers (offsets) and managed by a charity, the Great Barrier Reef Foundation.

The European Union offers support to farmers under the Common Agricultural Policy (CAP), where the pillar 2 measures include support for rural development, which can be used for agri-environmental schemes (Brady et al., 2021). However, it requires co-funding from Member States, and according to a report from the European Court of Auditors other more productivist priorities for support schemes tend to prevail, even where nitrogen leaching is causing serious challenges (ECA, 2016).

Economic instruments in terms of taxes on nitrogen have been practiced in European countries, where Austria, Finland and Sweden used to tax mineral nitrogen fertiliser. The fertiliser tax provided an incentive to reduce applications. In Finland 11% reductions in mineral fertiliser-N applications was obtained with a tax rate of EUR 0.44 per kg N, although study design does not make it possible to attribute this entirely to the tax (Rougoor, Van Zeijts, Hofreither & Bäckman, 2001). While Austria and Finland revoked their taxes upon becoming members of the EU in 1995, Sweden continued its scheme up to 2010. In a high-quality econometric study, Sweden's National Institute of Economic Research (KI, 2014) finds that the Swedish tax resulted in a net reduction of fertiliser-N use of about 6%, corresponding to about 10 000 tonnes of N annually - at a rate of SEK 1.80 (EUR 0.18) per kg N (Figure 5.2). For this tax rate the analysis finds that abatement costs per kg of N are as low as EUR 0.09 per kg N and competitive with most other measures. For higher abatement levels, the analysis shows that unit costs will increase but remain modest (up to EUR 0.6 per kg N). Mohlin, 2013 shows how the tax helped reduce $N_2O$ emissions as well. With the time series available (25 years) a long-run price elasticity of -0.4 was identified. In the United States most states have sales taxes on fertiliser to help finance inspections, but the rates are small and unlikely to affect behaviour (Sud, 2020).

## Figure 5.2. Price of mineral fertiliser (including tax) and use of nitrogen in Sweden.




Notes: N tax rate: EUR 0.18/kg N ; reduction in the use of N mineral fertilisers: 10 000 tonnes N/year (6%)
Source: KI, 2014.

Croatia, an EU Member State, has a tiny levy on mineral fertiliser at 3.70 Kuna (EUR 0.50) per metric tonne of N (UNECE, 2014; Hrvatske Vode, 2020). Belgium's region of Wallonia has an agricultural tax with a complex formulae including nitrogen in manure from livestock, with an effective tax rate of EUR 0.05 per kg N, although farms that have sufficient manure storage capacity are exempt (SPWARNE, 2017). France has a tax for which large livestock installations (>90 livestock units - LSU) are liable, with an annual rate of EUR 3 per LSU, corresponding to about EUR 0.03/kg N (Marcus and Simon, 2015). In Denmark a penalty of EUR 1.34/kg N applies for exceeding the N-quota, rising to EUR 2.68/kg N for the rare exceedances above 10 kg N/ha (Christensen, 2004).

Considering that environmental taxes should reflect the external costs at stake, some recent studies are suggesting that higher rates would be relevant. Keeler et al., 2016 estimate the external costs of nitrogen at up to USD 10/kg N for a US catchment, focusing on GHG, ammonia and drinking water. In a European study, the external costs of eutrophication per kg of surplus fertiliser-N leaching to a water body were found to average EUR 3 with a range from EUR 0.3-10, and rising to EUR 2-11.5 when factoring in cascading health burdens of ammonia and drinking water contamination plus GHG emissions (at EUR 24/t$CO_2$eq) (Andersen et al., 2019). The valuation of eutrophication refers to benefits to recreation and waterfront property owners, omitting biodiversity benefits per se, reflecting the challenges in estimating and capturing accurately the full range of externalities. Differences in population densities of catchments and the variability in leaching according to the site conditions of soil, hydrology etc. explain the range of estimations (see also Henryson, 2020).

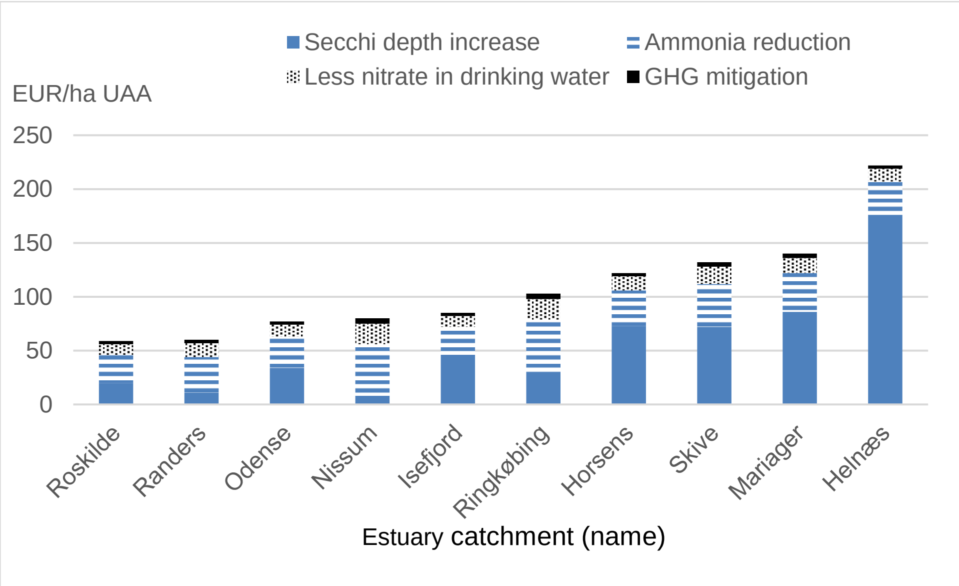
Since the externalities stem from the loss of nitrogen, rather than from its use, the most appropriate model for its taxation would be to target the nitrogen surplus, as first proposed by Hansen, 1999, rather than nitrogen inputs per se. This requires bookkeeping systems at farm level to account for the surplus. However the data will be readily available as well of the inputs from purchases of mineral fertilisers and fodder, as of the outputs from the sales of crops.[5] The optimal tax rate would ideally depend on the catchment specific circumstances. A uniform national tax rate on farm nitrogen surpluses might suffice to provide abatement incentives while being administratively feasible. Based on the estimates of externalities available (see above) the rate might start from EUR 1-2/kg N. It should be considered to allow a bottom deduction to reflect the share of organic nitrogen that cannot be readily utilised by plants within a 10-year timeframe, i.e. about 10% (or about 17-20 kg N/ha). There would be a need to analyse more carefully the distributional implications for different groups of farmers. To improve the acceptability of the tax for farmers, the revenues could be recycled back to the agricultural sector to lower other taxes, possibly with a share earmarked for investment in manure management technologies.

[5] For the nitrogen surplus accounting methodology see Eurostat 2013.

# 6 Policy opportunities for reaping the GHG benefits

With the modest shadow prices for the social cost of carbon applied in several countries (see OECD, 2018b), most opportunities for GHG reductions in agriculture will represent welfare economic costs. This observation holds even if the social cost of carbon is doubled. Inclusion of the ancillary benefits from reductions in nitrogen use, related to eutrophication ('secchi depth') as well as to human health effects of ammonia and ground/drinking water contamination, can however – despite the uncertainties involved – make a considerable difference as seen from Figure 6.1.

## Figure 6.1. Ancillary benefits increase the value of GHG mitigation




Note: Pricing GHGs at EUR 24/t$CO_2$eq. UAA = Utilised agricultural area.
Source: Andersen et al., 2019.

A recent analysis of seven measures to reduce agricultural GHGs estimates the ancillary benefits of reducing excess nitrogen between EUR 3.35 and EUR 8 per kg N (Dubgaard and Ståhl, 2018). Still, six of seven measures fail to return a net social benefit, with mitigation costs ranging from EUR 20 to 188 per tonne of $CO_2$-eq. The measures with ancillary benefits comprise biogas, nitrification inhibitors and grassland conversions of cultivated peatlands among others. The single measure returning a positive cost-benefit ratio is acidification of manure, which reduces methane and ammonia emissions by increasing the share of inorganic nitrogen in manure available for use as fertiliser. Still, this result hinges on farmers actually reducing their use of mineral fertilisers, despite not being legally required to do so.

Conversion of cultivated peatlands to permanent grasslands is one measure that has attracted considerable interest for its joint benefits, as perennial grasses both reduce nitrogen leaching and net GHG emissions. With regard to the latter, the peatlands would provide ecosystem services as sinks for carbon, but they would also emit more methane, so the net outcome hangs in the balance, whereby the economic value hinges on the availability of land use, land-use change and forestry (LULUCF) climate credits. In Denmark mainly the mosaic of smaller peatlands in watercourse valleys are candidates for conversion, as the larger, coherent peatland areas are in use for vegetable production, making them costly for conversion.

Grasslands can be used for provision of green protein to animal husbandry, but the processing would not be commercially viable unless extended value chains are mobilised. One option recently analysed is the combination of a grass refinery with biogas production. It would be sufficiently viable to attract investors, but hinging on the continuation of relatively high feed-in-tariffs for biogas. However, from a welfare economic perspective the results have been found to be negative, despite CAP support being factored in ex-ante. This result is largely due to environmental gains being too limited to justify the public financial support. The analysis priced nitrogen reductions at EUR 3.35 per kg surplus N (Martinsen and Andersen, 2020).

As shown in Chapter 3, there are numerous potential win-win measures to mitigate GHG emissions and nitrogen losses jointly, but one of the most reliable short-term measures remains a simple reduction in the use of excess mineral fertilisers by substitution with the already available manure, thus improving N use efficiency. Indeed, a literature review conducted for OECD suggest that measures based on improved fertiliser use efficiency are cost-effective in mitigating climate change (MacLeod et al., 2015). A recent study thus shows that the littoral states of the Baltic Sea could reduce mineral fertiliser use by 900 000 metric tonnes of nitrogen annually by diminishing over-fertilisation and substitute with manure-N (Herman and Tanzer, 2020), corresponding to GHG reductions of 9-15 million tonnes of $CO_2$eq, depending on the originating region of the averted mineral fertilisers.

The Court of Justice of the EU in its ruling C-543/16 (European Commission vs Germany) acknowledges that climatic conditions etc. may influence the surplus of nitrogen, but maintains that under the Nitrates Directive it is not permissible to legalise a nitrogen surplus, as it will violate fertilisation planning requirements (fertilisers to match crop needs etc.) and environmental objectives. Against the claims of the defendant that solid manure of ruminants etc. would not be prone to leaching, the Court maintains that restrictions are a legal obligation, since studies clearly demonstrate the mineralisation processes, also at lower temperatures (CJEU, 2018). Considering current nitrogen surpluses (Figure 2.2) fertilisation planning seems to deserve closer attention in several EU Member States.

With the EU's Farm-to-Fork strategy (EC, 2020a) that aims for a 50% reduction in nutrient surpluses while converting 25% of farmlands to organic agriculture, and with the prospects for the future CAP, there will be ample opportunity for Member States to attach strings to how public financial support to farmers is provided in the future. CAP payments will come to rely on elaboration of national CAP Strategic Plans (CSP), that while allowing Member States more leverage will be closely supervised by the European Commission (EC, 2019b). A particular emphasis can be expected on how the CSPs respond to the climate and environment objectives of the EU. Depending on the final outcome of the negotiations, 20-30% of the basic payments will have to be disbursed via dedicated Eco-schemes, while a comparable share of rural development funds should focus on activities of most direct value for the environment and climate. These payments will compensate farmers for entering into agri-environment-climate commitments of beneficial practices. It is to be expected that efforts will be stepped up as compared to the previous CAP requirements (e.g. for Ecological Focus Areas) that were criticised by the EU's Court of Auditors for promoting low-impact options mainly (ECA, 2020). There will be stronger conditionality measures to secure a basic level of environmental and climate management, e.g. to preserve and enhance carbon stocks in agricultural soils. However, CAP legislation remains not well connected to other EU strategies, and there is concern that Member States may favour business-as-usual approaches to how payments are made, without sufficient emphasis on more result-based options (Maréchal et al., 2020).

Moreover, with the plans to introduce a carbon border adjustment mechanism for certain imported (not yet identified) products (EC, 2020b; Dybka et al.,2020), the implications could be an increase in mineral fertiliser costs, which would make the utilisation of manure as fertiliser more attractive in EU Member States. Considering discrepancies in emission intensities of mineral fertiliser production in different world regions (see Chapter 4), applying the carbon border adjustment mechanism to mineral fertilisers would provide a financial incentive to reduce emissions for those producers that wish to maintain exports to the European Union (Andersen, 2018). Under WTO rules the mechanism would be permissible vis-à-vis jurisdictions without a carbon price, but should producer countries choose to implement comparable pricing mechanisms (China is expected to expand its ETS to eight industry sectors in 2021), the resulting price incentives would in any case help correcting and shifting the balance in fertiliser use away from mineral fertilisers and towards better use of manure. While the EU has already an anti-dumping duty on mineral fertilisers from Russia and United States that is WTO-compatible, a carbon-border adjustment mechanism will not go unchallenged (EC, 2019c).

# 7 Discussion and conclusions

A major challenge in controlling nitrogen are the cascading effects, where nutrients released into the environment can move along multiple pathways to various receptors. N exists in dissolved, particulate and gaseous forms and is highly labile (Shortle and Horan, 2017). Efforts to reduce nitrogen in one pathway can increase its flows in another, changing the location, type and timing of environmental burdens. As such its site-specificity differs fundamentally from that of $CO_2$ which is transported and diffused uniformly. This contradiction has implications for policy design, as for instance an emission trading system for nitrogen would be difficult to control and manage. Reductions in, say, leaching to water bodies could have been attained with measures transforming manure-N into ammonia, so that only a comprehensive coverage of all sources and emissions would qualify, which would be demanding to administrators. The site-specificity of nitrogen moreover implies that there are no exact marginal costs of nitrogen from which the rate of taxes on mineral fertiliser-N could be derived.

A further complication is that the nitrogen supported crop production is characterised by uncertain and random variations in soil and climate conditions that affect yields and leaching. The so-called ‘insurance application’ of fertilisers stems from these uncertainties, as it can be rational for farmers to apply extra nitrogen to minimise on their risks. While yield predictions and the derived fertiliser needs will often be based on efficiency assumptions, an ‘optimal’ climate rarely occurs. Farmers will in most years be applying too much fertiliser, which can be justified as economically optimal when the risks are factored in, if the cost of adding fertiliser is low compared to the value of the possible yields. The gain farmers will obtain can be further influenced by climate variability, when a rich harvest leads to declining market prices, whereby what under the circumstances should be the optimal fertiliser dose changes ex-post.

IPCC estimates with medium confidence a global GHG reduction potential of 30-710 million tonnes $CO_2$eq for cropland nutrient management relating to fertiliser applications (IPCC, 2019). It is not crystal clear how this estimate was derived from the references cited, nor the extent to which manure management is considered. Yet, if half the global mineral fertiliser use of 120 million tonnes-N (FAO data) is considered lost for crops as assumed by IPCC (2019) and reductions are targeted where producer emissions are highest (15 kg GHG emissions/kg N), there could be a global mitigation potential of 900 million tonnes $CO_2$eq. Substituting mineral fertiliser with manure-N could provide additional mitigation potential. Considering OECD countries with a total of 27 million tonnes manure-N and assuming a possible utilisation rate improvement from 25% to 80%, a further 15 million tonnes of mineral fertiliser-N use could be averted, yielding an additional mitigation potential of up to 220 million t$CO_2$eq. It will require a comprehensive transformation in farmers’ use of manure and a wide appreciation of the nitrogen mineralisation processes of solid dung. FAO, 2021 estimates global agricultural emissions at 5 300 million t$CO_2$eq, and thus a nitrogen fertiliser mitigation potential of up to 1 120 million t$CO_2$eq corresponds to about 20% of global agricultural emissions. Improved use of manure-N in China and other emerging economies could add to this potential. These figures make the potential of a European meat tax at EUR 60/t$CO_2$eq fade, as it would reduce GHG emissions by merely 32 million t$CO_2$eq by depressing consumption of ruminant meat by 15%, with a rebound of 1% and 7% for pig and poultry meat (Wirsenius et al., 2011).

For rapid and deep reductions to materialise, as stipulated under the UNFCCC Paris Agreement, a radical change is required in the approaches of public support to farmers. Public financial support should be redirected to encourage farmers to adopt best practices for manure management, including appropriate storage containers and spreading machinery and possible win-win practices (Brady et al., 2021). Equally important is knowledge dissemination and education of farmers to improve their understanding of how best to use fertilisers. A fertiliser plan should be compulsory for each farmer and bookkeeping of actual fertiliser use become mandatory, preferably with online reporting to underpin enforcement. Ideally, the nitrogen balance should be done at the plot level (the agronomic unit), as required by the Fertiliser Application Ordinance (DüV-20) in Germany since 2020 (Klages et al., 2020). Advisory services with a high professional standard that can provide support to individual farmers should be eligible for public financial support. With failures to meet nitrogen policy targets over the past decades and considering the urgency of reducing GHG emissions, quotas could be imposed on the use of mineral fertilisers to ensure that a balancing requirement (between crop needs and nutrient supply) is met. Such quotas would have to be differentiated according to crops and soils to take account of the different needs and circumstances of different types of farming. Maximum animal densities should be prescribed to avoid huge spatially concentrated volumes of manure. Fine levels for exceeding the requirements should be sufficient to cancel out any economic gains and constitute a real penalty, not an indulgence letter.

Levies or taxes on mineral fertilisers can further underpin efforts to reduce nitrogen and GHG emissions. The experience reviewed here shows that such taxes help reduce insurance fertilisation and can do so at low cost. Without it, transport of manure to fields over distances of more than 10-15 km will not be economically sound. By increasing the relative price of mineral fertilisers it will support the substitution with manure-N and alleviate the costs of proper handling. Farmers can be compensated through reductions in other taxes, e.g. land value taxes (Milne and Andersen, 2012). The most appropriate model is to tax the nitrogen surplus, rather than nitrogen inputs per se, based on bookkeeping at farm level. Further research would be required for designing schemes appropriate to national circumstances, while taking into account the data uncertainties.

# Annex A. Manure nitrogen utilisation rates in national codes of good agricultural practices of EU countries

## Figure A.1. Utilisation rates in % for manure-N in national codes of good agricultural practices

| EU member state[a] | Cattle[b] | | Pigs[b] | | Layer[b] | | Broiler[b] | Sheep[b] |
|---|---|---|---|---|---|---|---|---|
| | Slurry | Solid | Slurry | Solid | Slurry | Solid | | |
| Austria | 50 | 5/15 | 65 | 5/15 | 60 | 30 | 30 | n.r. |
| Belgium (Flanders) | 60 | 30 | 60 | 30 | 60 | 30 | 30 | 30 |
| Bulgaria | 20-35 | 20 | 40-45 | 20 | 40-50 | 40-50 | 40-50 | n.r. |
| Czech Republic | 60 | 40 | 60 | 40 | 60 | 40 | 40 | 40 |
| Denmark[c] | 70 | 65[d] | 75 | 65[d] | 70 | 65 | 65[d] | 65[d] |
| Estonia | 50 | 25 | 50 | 25 | 50 | 25 | 25 | 25 |
| France[e] | Low C:N | High C:N | Low C:N | High C:N | Low C:N | Low C:N | Low C:N | High C:N |
| Germany | 50 | 25-30 | 60 | 25-30 | 60 | 30 | 60 | n.r. |
| Greece | 20-35 | 10 | 25-45 | 10 | 20-30 | 20-30 | 20-30 | 10 |
| Ireland | 40 | 30 | 50 | 50 | 50 | 50 | 50 | n.r. |
| Italy[f] | 24-62 | 24-62 | 28-73 | 28-73 | 32-84 | 32-84 | 32-84 | n.r. |
| Latvia | 50 | 25 | 50 | 25 | 30 | 25 | 25 | n.r. |
| Lithuania | | 35[g] | | 35[g] | | | 35[g] | 35[g] |
| Luxembourg | 25-50 | 30-50 | 30-60 | 30-50 | n.r. | 50 | 50 | n.r. |
| Netherlands | 60 | 40 | 60-70 | 55 | 60/70 | 55 | 55 | n.r. |
| Poland | 50-60 | 30 | 50-60 | 30 | 50-60 | 30 | 30 | 30 |
| Portugal | 55-75 | 30-60 | 50-80 | 40-60 | 50-70 | 40-60 | 40-60 | 40-60 |
| Romania | 50 | 30 | 50 | 30 | n.r. | 30 | 50 | n.r. |
| Slovakia | 50 | 30 | 50 | 30 | 30 | 50 | 50 | n.r. |
| Slovenia | 50 | 30 | 50 | 30 | 30 | 50 | 50 | n.r. |
| Sweden | 40-50[h] | 36-41[h] | 57 | 47 | n.r. | 48 | 47/57[i] | 38 |
| United Kingdom[k] | 20/35[j] | 10 | 25/50[j] | 10 | n.r. | 20/35[j] | 20/30[j] | 10 |

Notes: a) n.r.= not reported. Belgium-Wallonia, Croatia, Cyprus, Finland, Hungary, Malta, and United Kingdom-Northern Ireland did not report utilisation rates; Spain: different values depending on the region.
b) Use of national coefficients could change the nitrogen balances, e.g. Denmark's nitrogen surplus is found to be 125 kg N/ha and 84 kg N/ha for 1995 and 2015 respectively, thus yielding a reduction of 41 kg N/ha. (Blicher-Mathiesen et al., 2019).
c) Also includes residual N effects in the following years after application.
d) 45% for deep litter.
e) C:N = carbon to nitrogen ratio.
f) According to soil type and time of application.
g) First year only. Total over 3 years is 70%.
h) Depending on class of animal.
i) Deep litter/other.
j) Autumn/spring application.
k) Membership ended by 2020.
Source: Webb et al., 2013.

# References

Andersen M.S. (2018), “Border Adjustment with Taxes or Allowances to Level the Price of Carbon”, in *Innovation Addressing Climate Change Challenges, Market-Based Perspectives*, Critical Issues in Environmental Taxation series, edited by Hymel M. et al., Edward Elgar Publishing.

Andersen M.S. (2002), ”Agriculture, Trade and Environment Linkages in the Pig Sector: A Comparative Study of Manure Regulations and their Significance for the Competitiveness of the Pig Sector”, paper presented to the 15th session of the OECD Joint Working Party on Agriculture and Environment (JWPAE), COM/AGR/CA/ENV/EPOC(2002)29.

Andersen M.S. et al. (2020), ”Governance of the Baltic Sea: Understanding Implementation Shortfalls in Nine Littoral States”, BONUS TOOLS2SEA DL1.1B, Aarhus University, https://projects.au.dk/bonus-tools2sea/publications/.

Andersen M.S., Levin G. and M.V. Odgaard (2019), ”Economic Benefits of Reducing Agricultural N Losses to Coastal Waters for Seaside Recreation and Real Estate Value in Denmark”, *Marine Pollution Bulletin*, 140, https://doi.org/10.1016/j.marpolbul.2019.01.010.

Aronsson H. et al. (2016), “The Ability of Cover Crops to Reduce Nitrogen and Phosphorus Losses from Arable Land in Southern Scandinavia and Finland”, *Journal of Soil and Water Conservation*, 71(1) , https://doi.org/10.2489/jswc.71.1.41.

Blicher-Mathiesen G. et al. (2019), *Landovervågningsoplande 2017*, Scientific Report 305, National Center for Environment and Energy (DCE), Aarhus University, https://dce2.au.dk/pub/SR305.pdf (in Danish).

Brady M. et al. (2021), “Strengthening the Policy Framework to Resolve Lax Implementation of the Baltic Sea Action Plan for Agriculture”, forthcoming in *Ambio*.

Brentrup F. et al. (2018), “Updated Carbon Footprint Values for Mineral Fertiliser from Different World Regions”, paper presented to the 11th International Conference on Life Cycle Assessment of Food (LCA Food), Bangkok, Thailand.

Brentrup F. and C. Palliere (2008), “GHG Emissions and Energy Efficiency in European Nitrogen Fertiliser Production and Use”, paper presented to the International Fertiliser Society, IFS Proceedings 639.

Cestti R., Srivastava J. and S. Jung (2003), “Agriculture Non-Point Source Pollution Control: Good Management Practices - The Chesapeake Bay Experience”. Working Paper 7, The World Bank, https://openknowledge.worldbank.org/handle/10986/15119.

Christensen B. (2004), ”Bøde ved overgødskning”, The Knowledge Service, University of Copenhagen, https://videntjenesten.ku.dk/raadgivning/spoergsmaal_og_svar/pyntegroent/b_de_ved_overg_dskning/ (in Danish).

CJEU (2018), “Judgment of the Court: Failure of a Member State to fulfill obligations - Directive 91/676/EEC”, Case C - 543/16 concerning an infringement suit versus Germany supported by Denmark, Court of Justice of the European Union, https://curia.europa.eu/juris/document/document.jsf?text=&docid=203231&pageIndex=0&doclang=DE&mode=req&dir=&occ=first&part=1.

Collins L. (2019), “Stiesdal to Produce Cheap Carbon-negative Aviation Fuel by 2025”, *Recharge*, 17 July, https://www.rechargenews.com/transition/stiesdal-to-produce-cheap-carbon-negative-aviation-fuel-by-2025/2-1-640010?%20Daily%20Newsletter=&utm_term=0_8680971c0c-ea1baf7cb3-19449777.

De Klein J.J.M. and A.K. van der Werf (2014), ”Balancing Carbon Sequestration and GHG Emissions in a Constructed Wetland”, *Ecological Engineering*, 66, https://doi.org/10.1016/j.ecoleng.2013.04.060.

Diaz R.J. and R. Rosenberg (2008), “Spreading Dead Zones and Consequences for Marine Ecosystems”, *Science*, 321(5891), https://doi.org/10.1126/science.1156401.

Dubgaard A. and L. Ståhl (2018), ”Omkostninger ved virkemidler til reduktion af landbrugets drivhusgasemissioner: Opgjort i relation til EU’s 2030-målsætning for det ikke-kvotebelagte område”, Department of Food and Resource Economics (IFRO), Report 271, University of Copenhagen, https://static-curis.ku.dk/portal/files/204121155/IFRO_Rapport_271.pdf (in Danish).

Duhon M., McDonald H. and S. Kerr (2015), ”Nitrogen Trading in Lake Taupo: An Analysis and Evaluation of an Innovative Water Management Policy”, Motu Working Paper 15-07, Motu Economic and Public Policy Research, Wellington, http://dx.doi.org/10.2139/ssrn.2653472.

Dybka D. et al. (2020), “Border Carbon Adjustments in the EU: Issues and Options”, European Roundtable on Climate Change and Sustainable Transition, https://ercst.org/border-carbon-adjustments-in-the-eu-issues-and-options/.

EC (2020a), *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions: A Farm to Fork Strategy - For a Fair, Healthy and Environmentally-friendly Food System*, COM(2020) 381 final, European Commission, Brussels, https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52020DC0381.

EC (2020b), “Carbon Border Adjustment Mechanism”, Legislative Proposal, Inception Impact Assessment, Directorate-General for Taxation and Customs Union, https://ec.europa.eu/info/law/better-regulation/have-your-say/initiatives/12228-Carbon-Border-Adjustment-Mechanism.

EC (2019a), “Summary Report on the Implementation of Direct Payments [except Greening] - Claim Year 2017”, https://ec.europa.eu/info/sites/default/files/food-farming-fisheries/key_policies/documents/summary-report-implementation-direct-payements-claim-2017.pdf.

EC (2019b), *The Post-2020 Common Agricultural Policy: Environmental Benefits and Simplification*, Directorate-General for Agriculture and Rural Development, https://ec.europa.eu/info/sites/info/files/food-farming-fisheries/key_policies/documents/cap-post-2020-environ-benefits-simplification_en.pdf.

EC (2019c), *Commission Implementing Regulation (EU) 2019/576 of 10 April 2019 imposing a Provisional Anti-dumping Duty on Imports of Mixtures of Urea and Ammonium Nitrate originating in Russia, Trinidad and Tobago and the United States of America*, http://data.europa.eu/eli/reg_impl/2019/576/oj.

EC (2018a), “Annexes to the Report from the Commission to the European Parliament and the Council – 11th Financial Report from the Commission to the European Parliament and the Council on the European Agricultural Guarantee Fund – 2017 Financial Year”, COM(2018) 628 final, https://eur-lex.europa.eu/resource.html?uri=cellar:68fb4e01-b804-11e8-99ee-01aa75ed71a1.0020.02/DOC_2&format=PDF.

EC (2018b), “Report from the Commission to the Council and the European Parliament on the Implementation of Council Directive 91/676/EEC concerning the Protection of Waters against Pollution caused by Nitrates from Agricultural Sources based on Member State reports for the period 2012-2015”, COM(2018) 257 final, https://ec.europa.eu/environment/water/water-nitrates/pdf/nitrates_directive_implementation_report.pdf.

ECA (2020), *Biodiversity on Farmland: CAP Contribution Has Not Halted the Decline*, Special Report 13. European Court of Auditors, https://www.eca.europa.eu/Lists/ECADocuments/SR20_13/SR_Biodiversity_on_farmland_EN.pdf.

ECA (2016), *Combating Eutrophication in the Baltic Sea: Further and More Effective Action Needed*, Special Report 3, https://www.eca.europa.eu/Lists/ECADocuments/SR16_03/SR_BALTIC_EN.pdf.

Erisman J. W. et al. (2008), "How a Century of Ammonia Synthesis Changed the World", *Nature Geoscience*, 1, October 2008, https://doi.org/10.1038/ngeo325.

Eurostat (2017), "Glossary: Livestock Units (LSU)", *Statistics Explained*, https://ec.europa.eu/eurostat/statistics-explained/pdfscache/1246.pdf.

Eurostat (2013), *Nutrient Budgets – Methodology and Handbook*, Version 1.02, Eurostat and OECD, Luxembourg, https://ec.europa.eu/eurostat/documents/2393397/2518760/Nutrient_Budgets_Handbook_%28CPSA_AE_109%29_corrected3.pdf/4a3647de-da73-4d23-b94b-e2b23844dc31.

FAO (2021), *Emissions due to Agriculture. Global, Regional and Country Trends 2000–2018,* FAOSTAT Analytical Brief Series No 18, Food and Agriculture Organization of the United Nation, Rome, https://www.fao.org/publications/card/fr/c/CB3808EN/.

Grant R. et al. (2011), *Landovervågningsoplande 2010*, Scientific Report 3, National Center for Environment and Energy (DCE), Aarhus University, www.dmu.dk/Pub/SR3.pdf (in Danish).

Hansen L.G. (1999), "A Deposit-Refund System Applied to Nitrogen Emissions from Agriculture", *Environmental Economics and Policy Studies*, 2, https://mpra.ub.uni-muenchen.de/26664/1/MPRA_paper_26664.pdf.

Hasler K. et al. (2017), "Eco-innovations in the German Fertilizer Supply Chain: Impact on the Carbon Footprint of Fertilizers", *Plant Soil Environ.*, 63(12), https://doi.org/10.17221/499/2017-PSE.

Hellsten S. (2017), *Ammonia Emissions in Sweden: Inventories, Projections and Potential for Reduction*, Report C237, Swedish Environmental Research Institute (IVL), Stockholm, https://www.ivl.se/download/18.1369484715f59ce4bab256e/1513773189822/C237.pdf.

Henderson B. and J. Lankoski (2019), "Evaluating the Environmental Impact of Agricultural Policies", *OECD Food, Agriculture and Fisheries Papers*, 130, OECD Publishing, Paris, https://doi.org/10.1787/add0f27c-en.

Henryson K. (2019), "Modelling Site-dependent Environmental Impacts of Nitrogen Fertiliser Use in Life Cycle Assessments of Crop Cultivation", PhD thesis, Swedish University of Agricultural Sciences, Uppsala, https://pub.epsilon.slu.se/16426/7/henryson_k_191106.pdf.

Hermann L. and J. Tanzer (2020), "Nutrient Flows in the Baltic Sea Region", presentation at Baltic Marine Environment Protection Commission (HELCOM) Workshop on Nutrient Recycling Measures, 5 February, Helsinki, https://portal.helcom.fi/meetings/Nutrient%20recycling%20measures%20WS-699/Presentations/1_Proman_nutrient_flows.pdf.

Hrvatske Vode (Croatian Waters) (2020), Personal communication.

Huygens D. et al. (2020), "Technical Proposals for the Safe Use of Processed Manure above the Threshold established for Nitrate Vulnerable Zones by the Nitrates Directive (91/676/EEC)", European Commission's Joint Research Centre, *JRC Science for Policy Report*, JRC121636, EUR 30363 EN, Publications Office of the European Union, Luxembourg, https://ec.europa.eu/jrc/en/publication/eur-scientific-and-technical-research-reports/technical-proposals-safe-use-processed-manure-above-threshold-established-nitrate-vulnerable.

Ingelsson M. and L. Drake (1998), ”Price Elasticity of Nitrogen Fertilisers in Sweden”, *Swedish Journal of Agricultural Research*, 28(4).

IPPC (2019), *Climate Change and Land*, an IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse Gas Fluxes in Terrestrial Ecosystems, the Intergovernmental Panel on Climate Change, Geneva, https://www.ipcc.ch/site/assets/uploads/2019/08/Fullreport-1.pdf.

Jensen K. et al. (1994), ”Udbringningsudstyrs betydning for udnyttelse af naturgødning”, in Andersen M.S. (ed.), *Spredningen af renere teknologi i landbruget*, Working report from the Danish Environmental Protection Agency, 59, https://www2.mst.dk/Udgiv/publikationer/1994/87-7810-269-3/pdf/87-7810-269-3.pdf (in Danish).

Keeler B.L. et al. (2016), “The Social Costs of Nitrogen”, *Science Advances*, 2(10), https://doi.org/10.1126/sciadv.1600219.

Kemp R. and J. Rotmans (2004), “Managing the Transition to Sustainable Mobility”, in *System Innovation and the Transition to Mobility*, *Theory, Evidence and Policy*, edited by Elzen B., Geels F. W. and K. Green, Edward Elgar Publishing.

KI (2014), *Miljö, ekonomi och politik 2014*, Konjunkturinstitutet (National Institute of Economic Research), Stockholm, https://www.konj.se/download/18.42684e214e71a39d0722ed0/1436516834703/Milj%C3%B6+ekonomi+och+politik+2014.pdf (in Danish).

Klages S. et al. (2020), “Nitrogen Surplus—A Unified Indicator for Water Pollution in Europe?”, *Water*, 12(4), http://dx.doi.org/10.3390/w12041197.

Leonhard A. (2019), “Skyclean Inventor says Agri-waste-to-bio-jet-fuel Tech could be Denmark's Big Export Boom”, *Energy Watch*, 4 November, https://energywatch.eu/EnergyNews/Cleantech/article11732650.ece.

Lötjönen S., Temmes E. and M. Ollikainen (2020), “Dairy Farm Management when Nutrient Runoff and Climate Emissions Count”, *American Journal of Agricultural Economics*, 102(3), https://doi.org/10.1002/ajae.12003.

MacLeod M. et al. (2015), “Cost-Effectiveness of Greenhouse Gas Mitigation Measures for Agriculture: A Literature Review”, *OECD Food, Agriculture and Fisheries Papers*, 89, OECD Publishing, Paris, https://doi.org/10.1787/5jrvvkq900vj-en.

Marcus V. and O. Simon (2015), « Les pollutions par les engrais azotés et les produits phytosanitaires - coûts et solutions », *Études et documents*, 136, Commissariat Général au Développement Durable (CGDD), Ministère de l'Écologie, du Développement durable et de l’Énergie, http://temis.documentation.developpement-durable.gouv.fr/docs/Temis/0083/Temis-0083486/22321.pdf.

Maréchal A. et al. (2020), “Aligning the Post-2020 Common Agricultural Policy with the European Green Deal”, Institute for European Environmental Policy (IEEP), September, https://ieep.eu/uploads/articles/attachments/34ac0c06-a1f4-4a72-87bb-7d548956b107/Aligning%20the%20post-2020%20CAP%20with%20the%20Green%20Deal.pdf?v=63768770228.

Martinsen L. and M.S. Andersen (2020), *Environmental and Socio-economic Analysis of Integrated Grass Biorefinery Scenarios*, Scientific Report 383, Danish Centre for Environment and Energy (DCE), Aarhus University, http://dce2.au.dk/pub/SR383.pdf.

Milne J.E. and M.S. Andersen (2012), *Handbook of Research on Environmental Taxation*, Edward Elgar Publishing.

Mitchell I. and A. Baker (2019), “New Estimates of EU Agricultural Support: An “Un-Common” Agricultural Policy”, Center for Global Development, https://www.cgdev.org/sites/default/files/Mitchell-EU-Ag-Subsidies-Final_0.pdf.

Mohlin K. (2013), “Essays on Environmental Taxation and Climate Policy”, *Economic Studies*, 214, University of Gothenburg, https://gupea.ub.gu.se/bitstream/2077/33425/3/gupea_2077_33425_3.pdf.

Nielsen H.Ø. (2009), *Bounded Rationality in Decision-making, How Cognitive Shortcuts and Professional Values May Interfere with Market-based Regulation*, Issues in Environmental Politics series, Manchester University Press.

Nielsen O.K. et al. (2019), *Annual Danish Informative Inventory Report to UNECE: Emission Inventories from the Base Year of the Protocols to Year 2017*, Scientific Report 313, Danish Centre for Environment and Energy (DCE), Aarhus University, http://dce2.au.dk/pub/SR313.pdf.

OECD (2019), *Trends and Drivers of Agri-environmental Performance in OECD Countries*, OECD Publishing, Paris, https://doi.org/10.1787/b59b1142-en.

OECD (2018a), *Human Acceleration of the Nitrogen Cycle: Managing Risks and Uncertainty*, OECD Publishing, Paris, https://doi.org/10.1787/9789264307438-en.

OECD (2018b), *Cost-Benefit Analysis and the Environment: Further Developments and Policy Use*, OECD Publishing, Paris, https://doi.org/10.1787/9789264085169-en.

OECD (2017), *OECD Environmental Performance Reviews: New Zealand 2017*, OECD Environmental Performance Reviews, OECD Publishing, Paris, https://doi.org/10.1787/9789264268203-en.

OECD (2015a), "The Lake Taupo Nitrogen Market in New Zealand: A Review for Policy Makers", *OECD Environment Policy Papers*, No. 4, OECD Publishing, Paris, https://doi.org/10.1787/5jrtg1l3p9mr-en.

OECD (2015b), *Innovation, Agricultural Productivity and Sustainability in Canada*, OECD Food and Agricultural Reviews, OECD Publishing, Paris, https://doi.org/10.1787/9789264238541-en.

OECD (2012), *Water Quality and Agriculture: Meeting the Policy Challenge*, OECD Studies on Water, OECD Publishing, Paris, https://doi.org/10.1787/9789264168060-en.

OECD (2006), *OECD Environmental Performance Reviews: United States 2005*, OECD Environmental Performance Reviews, OECD Publishing, Paris, https://doi.org/10.1787/9789264013179-en.

Olesen J.E. et al. (2020), *Opdatering af kvælstofudvaskning fra økologiske bedrifter*, Report 176, National Center for Food and Agriculture (DCA), Aarhus University, https://dcapub.au.dk/djfpublikation/djfpdf/DCArapport176.pdf (in Danish).

Pedersen A.B. et al. (2012), ”Optimising the Effect of Policy Instruments: a Study of Farmers' Decision Rationales and How they Match the Incentives in Danish Pesticide Policy”, *Journal of Environmental Planning and Management*, 55(8), https://doi.org/10.1080/09640568.2011.636568.

Rougoor C.W. et al. (2001), “Experiences with Fertilizer Taxes in Europe”, *Journal of Environmental Planning and Management*, 44(6), https://doi.org/10.1080/09640560120087615.

Schullehner J. et al. (2018), “Nitrate in Drinking Water and Colorectal Cancer Risk: A Nationwide Population-based Cohort Study”, *Int. J. Cancer*, 143(1), https://doi.org/10.1002/ijc.31306.

SEGES (2018), *Pilotprojektordningen om præcisionslandbrug, Erfaringer fra konsulentordningen*, Aarhus, https://lbst.dk/fileadmin/user_upload/NaturErhverv/Filer/Tvaergaaende/Praecisionslandbrug/SEGES_-_foerste_aars_erfaringer.pdf (in Danish).

Schroder J.J. (2014), “The Position of Mineral Nitrogen Fertilizer in Efficient Use of Nitrogen and Land: A Review”, *Natural Resources*, 5(15), https://doi.org/10.4236/nr.2014.515080.

Shortle J. and R.D. Horan (2017), ”Nutrient Pollution: A Wicked Challenge for Economic Instruments”, *Water Economics and Policy*, 3(2), https://doi.org/10.1142/S2382624X16500338.

Simonis U.E. (1988), *Präventive Umweltpolitik*, edited by Berlin Social Science Center (WZB), Campus (in German).

Smith P. et al. (2020), “Canada’s Environmental Farm Plan: Evaluating Implementation, Use of Services and the Influence of Social Factors”, *Sustainable Agriculture Research*, 9(4), https://doi.org/10.5539/sar.v9n4p1.

SPWARNE (2017), « La taxe sur les charges environnementales générées par les exploitations agricoles », Service Public de Wallonie Agriculture, Ressources Naturelles et Environnement, https://agriculture.wallonie.be/la-taxe-sur-les-charges-environnementales-generees-par-les-exploitations-agricoles.

Sud M. (2020), “Managing the Biodiversity Impacts of Fertiliser and Pesticide Use: Overview and Insights from Trends and Policies across Selected OECD Countries”, *OECD Environment Working Papers*, 155, OECD Publishing, Paris, https://doi.org/10.1787/63942249-en.

Svanbäck A. et al. (2019), Reducing Agricultural Nutrient Surpluses in a Large Catchment - Links to Livestock Density, *Science of the Total Environment*, 648, https://doi.org/10.1016/j.scitotenv.2018.08.194.

Petersen J and P. Sørensen (2008), *Gødningsvirkning af kvælstof i husdyrgødning – Grundlag for fastlæggelse af substitutionskrav*, The Faculty of Agricultural Sciences (DJF), Land Use 138, Aarhus University, https://pure.au.dk/portal/files/2428334/djfma138.pdf.

Sørensen P., Thomsen I.K. and J. J. Schröder (2017), ”Empirical Model for Mineralisation of Manure Nitrogen in Soil”, *Soil Research*, 55(6), https://doi.org/10.1071/SR17018.

Thorsøe M.H. et al. (2021), ”Promise and Performance of Agricultural Nutrient Management: Lessons from the Baltic Sea”, forthcoming in *Ambio*.

Tybirk K. et al. (2013), *Sustainable Manure Management in the Baltic Sea Region, Results, Cases and Project Recommendations*, Baltic Manure project co-financed by the EU (Baltic Forum for Innovative Technologies for Sustainable Manure Management), http://balticslurry.eu/wp-content/uploads/2016/06/Baltic-Manure-Final-report-2013.pdf.

UNECE (2014), *Environmental Performance Reviews, Croatia*, Second Review, Environmental Performance Reviews Series 40, United Nations Economic Commission for Europe, Geneva, https://www.unece.org/fileadmin/DAM/env/epr/epr_studies/ECE_CEP_172_En.pdf.

USEPA (2020), “State Progress Toward Developing Numeric Nutrient Water Quality Criteria for Nitrogen and Phosphorus”, United States Environmental Protection Agency, https://www.epa.gov/nutrient-policy-data/state-progress-toward-developing-numeric-nutrient-water-quality-criteria.

USEPA (1998), *National Strategy for the Development of Regional Nutrient Criteria*, Office of Water, 4304, EPA 822-R-98-002, https://www.epa.gov/sites/production/files/2018-10/documents/nutrient-strategy-1998.pdf.

Vollenweider R.A. (1968), “Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication”, Technical report DAS/CSI/68.27, OECD, Paris.

Webb J. et al. (2013), "An Assessment of the Variation of Manure Nitrogen Efficiency throughout Europe and an Appraisal of Means to Increase Manure-N Efficiency", *Advances in Agronomy*, 119, https://doi.org/10.1016/B978-0-12-407247-3.00007-X.

Wirsenius S., Hedenus F. and K. Mohlin (2011), "Greenhouse Gas Taxes on Animal Food Products: Rationale, Tax Scheme and Climate Mitigation Effects", *Climatic Change*, 108, https://doi.org/10.1007/s10584-010-9971-x.

WRI and WPCSD (2014), *Greenhouse Gas Protocol, GHG Protocol Agricultural Guidance, Interpreting the Corporate Accounting and Reporting Standard for the Agricultural Sector*, World Resources Institute, World Business Council for Sustainable Development, https://ghgprotocol.org/agriculture-guidance.