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Nitrogen deposition in the Nordic countries

Results from two research projects focused on Nordic nature and air pollution

Report prepared by:
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Results from two research projects focused on Nordic nature and air pollution

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Introduction

This report presents the results of two research projects, both funded by the Nordic Council of Ministers working groups, and both lead by Aarhus University, Department of Environmental Science. The projects have investigated two different aspects of nitrogen deposition to terrestrial nature areas in the Nordic countries and the results and methodological advances complement each other very well, providing recommendations for Nordic policy development for air pollution, especially with respect to emissions of ammonia.

The aim of the *Nordic Nature & Nitrogen* project was to update the regional air pollution models used in Denmark, Norway and Sweden with a more comprehensive parameterisation of the process of ammonia deposition, and investigate the consequences of this update on the distribution of nitrogen deposition on sensitive nature areas and the impact on biodiversity. The project was carried out as a collaboration between Department of Environmental Science (Aarhus University), the Norwegian Meteorological Institute and the Swedish Meteorological and Hydrological Institute.

The aim of the *Benefit Nature* project was to establish a methodology for development of a new system — Economic Valuation of Air pollution for nature (EVA-Nature) — to assess socio-economic benefits of reducing reactive nitrogen deposition to sensitive terrestrial nature areas in the global North. The project was carried out in a collaboration between Department of Environmental Science (Aarhus University), Department of Food and Resource Economics (University of Copenhagen), KM Miljø- og ressursøkonomi and Agricultural University of Iceland.

Research recommendations from the two projects

Despite long-term awareness, deposition of reactive nitrogen remains an important environmental issue relevant to the Nordic countries. Although the emissions of ammonia and nitrogen oxides, which contribute to the deposition of reactive nitrogen, have been declining over the last decades, the deposition of nitrogen is not decreasing at the same pace.

Nature areas experience a short delay before showing negative impacts on plant species richness, when nitrogen deposition increases, but face a much longer delay in recovery of plant species richness, when nitrogen deposition decreases. Combined with the slowing decline in deposition, this calls for attention when trying to anticipate, how new legislation for emissions impacts the preservation state of sensitive terrestrial nature.

As air pollution models move toward higher spatial resolution, we need to reevaluate the parameterisations of dry deposition. The parameters included in coarser regional models become less adequate at higher resolution, where parameters on small-scale surface differences are more important. More research, additional measurements and collaboration with the researchers who perform measurements of deposition fluxes is therefore necessary.

There is existing knowledge and analyses of the relationship between nitrogen deposition and plant species richness, but with a geographic bias towards the ecosystems of the UK, due to the many studies in this region. Even though UK nature resembles southern Nordic nature areas, it will be very beneficial to perform new analyses of the temporal trends in high-resolution nitrogen deposition, and plant species richness data for sensitive terrestrial nature areas in the Nordic countries.

Although other researchers have tried to develop an impact-pathway assessment methodology for assessing the socio-economic benefits of reducing reactive nitrogen deposition in nature areas, the valuation review highlights the current shortage of studies that apply appropriate methodologies to assess the environmental effects. Therefore, we recommend that new primary valuation studies pertaining to this specific pollution problem are initiated.

We have developed a framework for inclusion of costs of plant biodiversity loss from air pollution once such studies are available. Like other valuation estimates, there is a need to update them continuously, as values develop over time, not only due to changes in the natural environment, but also in society’s assessment of their importance.

The *Nordic Nature & Nitrogen* project

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This research project has revolved around the inclusion of the *bidirectional flux process* in 3D air pollution models that are developed within research groups in Denmark, Norway and Sweden and commonly applied for e.g. assessments of air quality and nitrogen deposition in relation to policy advice. The three air pollution models are:

- The Danish Eulerian Hemispheric Model – DEHM (Christensen, 1997; Frohn et al., 2002; Brandt et al., 2012) – developed at the Department of Environmental Science, Aarhus University.
- The European Monitoring and Evaluation Programme Model – EMEP MSC-W (Simpson et al., 2012; 2014; 2024 and references therein) – developed at the Norwegian Meteorological Institute.
- The Multi-scale Atmospheric Transport and Chemistry model – MATCH (Robertson et al., 1999; 2023) – developed at the Swedish Meteorological and Hydrological Institute.

The bidirectional flux characterizes the dynamic exchange of atmospheric ammonia (NH_3) between the atmosphere and the earth surface (especially vegetation), where vegetation acts simultaneously as both a source and a sink. Though it is challenging to model, this interaction between vegetation and air should be incorporated into air pollution models to accurately represent their complex relationship. NH_3 is emitted primarily from agricultural activities, and it is a very important component contributing to the total atmospheric reactive nitrogen (Nr) deposition, which again has an impact on vegetation and is of special importance in areas with nature sensitive to elevated levels of nutrients.

Within the Nordic Nature and Nitrogen project, three model groups have been working with the implementation of the bidirectional flux in their respective models, and compared the resulting concentrations of NH_3 and other Nr components to measurements in ambient air. Denmark has been selected as a case-study area and the differences in deposition distribution and quantity between the original (denoted **noBD**) and the bidirectional flux (denoted **BD**) versions of the models have been investigated for all three models across the Danish area.

The bidirectional flux parameterisation that has been used in the models, was originally developed in the Netherlands by Roy Wichink-Kruit (Wichink-Kruit et al., 2010; 2012; 2017) and later updated by Sebastiaan Hazelhorst (Hazelhorst, 2019) and the two have provided expert advice and guidance to the project group during the course of the project.

Methodology

This section provides a brief overview of the three air pollution models included in the project, together with a description of the bidirectional flux parameterisation (and the differences to the original dry deposition parameterisation).

The three air pollution models included in the project, all use mathematical and numerical techniques to simulate the chemical and physical processes that influence air pollutants. Based on information about meteorological parameters, land surface characteristics and emissions from anthropogenic and natural sources, dispersion, transport and transformation of air pollutants in the atmosphere can be predicted in 3D space and time, for a geographical area covering e.g. Europe and/or the Nordic countries. Many processes must be described in the models to account for all transformations, including dry and wet deposition, which describe the transfer of gases and particles from the atmosphere to the ground, either directly or through precipitation. The governing equations for these deposition processes are not directly known, and therefore it is necessary to parameterise the processes, i.e. try to describe what is going on with the aid of measurements of relevant parameters.

Dry deposition depends on several factors, including the amount of turbulence in the air, the roughness of the surface, the vegetation type and height, the time of day and year (due to seasonal and diurnal changes in vegetation cover and properties), the physical and chemical properties of the substance being deposited and the moisture content of the surface. In the models, the deposition is calculated using the concentration in the air, and a deposition velocity to give the downward flux. The modelled deposition velocity must account for all the above listed parameters influencing deposition. Several parameterisations have been developed for this purpose, the most popular in regional-scale air pollution models being the *resistance model* (see e.g. Seinfeld & Pandis, 1998). All three models in this project use this particular approach for calculating dry deposition velocities of NH_3 and other chemical components.

The *resistance model* represents the process of dry deposition in three steps, each contributing to the deposition velocity:

- The process of aerodynamic transport down through the atmospheric surface layer to a very thin layer of stagnant air just adjacent to the surface,
- the process of molecular (for gases) or Brownian motion (for particles) transport across this thin stagnant layer of air called the quasi-laminar sublayer, to the surface itself, and
- the process of uptake at the surface.

To calculate the deposition velocity, this modelling approach describes the three above steps as a series of resistances to deposition, analogous to how electrical resistances are combined in circuits. For dry deposition, the "resistances" corresponding to the above steps of deposition are:

- The *aerodynamic resistance,* r_a, depending on the turbulence intensity, which in turn depends on the stability of the lower atmosphere and the surface roughness,
- the *quasi-laminar resistance,* r_b, depending on the molecular properties of the depositing component and the characteristics of the surface, and
- the *surface* or *canopy resistance,* r_c. For vegetation, depending on the accessibility of the gas to reaction sites within the plant, and for soil, water and snow surfaces, depending on the reactivity and solubility of the gas as well as the moisture level and pH of the surface.

The *surface resistance,* r_c, is often comprised of several components, with one related to the uptake of gases through the stomatal openings of plants, and one or more related to uptake via other parts of vegetation (e.g. exterior on leaves) or soil/water surfaces (together often referred to as the non-stomatal pathway). When r_a, r_b and r_c are known, the deposition velocity, v_d, can be calculated as $v_d = 1/(r_a+r_b+r_c)$.

Most air pollution models using the resistance method for calculating the dry deposition, assume that the flux of NH_3 is only one-way; from the atmosphere to the ground. However, NH_3 is also released from the plant canopy, and when sources (concentrations of ammonium in liquid components) are sufficiently high, these emissions can exceed the downward flux, and a net emission can be observed. The point at which this occurs is known as the compensation point (CP). When the concentration of ambient NH_3 exceeds the CP, a downward flux is observed. When ambient NH_3 is below the CP, an emission is observed. We refer to this system as *bidirectional*.

The bidirectional flux has been part of the Dutch local scale NH_3 model (OPS, Wichink-Kruit et al., 2017 and references therein) for several years. In the Nordic Nature & Nitrogen project, the source code for the bidirectional flux parameterisation and the documentation of the underlying physics, methodology and parameters, have been shared with the project group and included in the three models DEHM, EMEP and MATCH. Prior to implementation, an analysis of the model characteristics and parameterisations between the three models was carried out. The result is shown in Table 1.

Table 1: Input data and parameterisations for compensation point implementation in DEHM, EMEP and MATCH.

	DEHM	**EMEP**	**MATCH**
Meteorology	WRF with grid-nudging	ECMWF	ECMWF-IFS
Horizontal resolution	150 km / 50 km / 16.67km / 5.56 km	0.1°× 0.1°/ 0.3°× 0.2°	0.2° × 0.2°/ 0.1° × 0.1°/ 5km × 5km
Vertical layers of CTM	29 sigma levels	20 hybrid levels of the met. model layers	50 hybrid levels of the met. model, reduced to 25 levels in MATCH
Vertical extent of CTM	100 hPa	100 hPa	~6-8 km
Surface concentration height	Lowest layer ~23 m, mid ~11m. Possible to downscale to 3 m.	Downscaled to 3 m	Downscaled to 3 m
Land-use database	Olsson World Ecosystem classes Version 1.4D	CCE/SEI for Europe, elsewhere GLC2000/CLM	CCE/SEI for Europe
Dry deposition	Resistance model for gases and aerosols (Emberson et al. 2000a, b; Venkatram and	Resistance model for gases and aerosols (Emberson et al 2000a, b; Venkatram and	Resistance model for gases (similar to Simpson et al., 2012). Aerosol deposition

	Pleim, 1999; Simpson et al., 2012)	Pleim, 1999; Simpson et al., 2012)	based on Simpson et al., 2012
Ammonia compensation point	None	None, but no dry deposition of NH_3 over growing crops	None, but no dry deposition of NH_3 to fertilized land use categories during the growing season
Stomatal resistance	DO3SE-EMEP: Emberson et al. 2000a, b, Tuovinen et al. 2004, Simpson et al 2012	DO3SE-EMEP: Emberson et al 2000a, b, Tuovinen et al 2004, Simpson et al 2012	Based on Simpson et al., 2012
Wet deposition - scavenging of gases	In-cloud scavenging of gases depend on meteorology and is based on Henry's law equilibrium. Different scavenging ratios are used for in-cloud and sub-cloud processes.	Scavenging calculated from the gas mixing ratio, precipitation rate and species-specific scavenging ratios. Different scavenging ratios are used for in-cloud and sub-cloud processes. Simpson et al 2012.	Wet scavenging is assumed proportional to precipitation intensity. Species specific scavenging coefficients are used, different for in-cloud and sub-cloud processes.
Wet deposition - scavenging of particles	In-cloud scavenging proportional to the fraction of cloud-water that hits the ground as precipitation. All particulate sulphate inside clouds assumed to be dissolved in droplets. Sub-cloud scavenging calculated from particle mixing ratio, precipitation rate, raindrop fall speed and size-dependent collection efficiency.	In-cloud: as gas scavenging above. Sub-cloud: Scavenging calculated from the particle mixing ratio, precipitation rate, raindrop fall speed and a size-dependent collection efficiency. Simpson et al., 2012	Precipitation intensity dependent scavenging coefficients for most particle components
Gas-phase chemistry	Strand & Hov 1993, extended with ammonia chemistry, secondary organic aerosols (Simpson et al 2012) and terpenes.	EmChem19 (Bergström et al., 2022; Simpson et al 2020)	Mainly based on EmChem09 (Simpson et al., 2012) with modified isoprene chemistry (Carter 1996, Langner et al., 1998); reaction rates updated, similar to EmChem19 (Bergström et al., 2022)
Cloud chemistry	Aqueous SO_2 chemistry	Aqueous SO_2 chemistry	Aqueous SO_2 chemistry
Coarse nitrate	Transfer of HNO_3(g) to aerosol NO_2 (Strand & Hov 1993) and distribution into coarse and fine fraction (PM_{10} and $PM_{2.5}$)	Transfer of HNO_3(g) to aerosol NO_3 on coarse dust and sea-salt, see Stadtler et al., 2018	Transfer of HNO_3(g) to aerosol NO_3 using rate from Strand & Hov 1993
NH_4NO_3 equilibrium	RH and T-dependent equilibrium coefficient	ISORROPIA-Lite (Kavakas et al., 2022)	RH- and T-dependent equilibrium coefficient (Mozurkewich 1993)

Setup for model calculations

The MATCH and DEHM models have in this project been set up to perform calculations with high resolution over Denmark. The DEHM model includes high-resolution land cover data for Denmark (Levin et al., 2022), with details of the location of sensitive nature areas. The high resolution over Denmark was chosen because Denmark is the Nordic country with the highest emissions of NH_3 to the air from anthropogenic activities. The MATCH model was run at three different scales: 1) A European domain with simulations at 0.2° × 0.2° resolution [Lat 30–72°N, Long 15°W–45°E], 2) nested simulations at 0.1° × 0.1° resolution covering Denmark, the Netherlands, Southern Sweden and Norway, and parts of neighboring countries, and 3) high-resolution simulations for Sweden with a spatial resolution of 5 km × 5 km. The Sweden-scale simulations will be reported elsewhere and are not included in this report. The DEHM model is run for the Northern Hemisphere with four nested domains over Europe, Northern Europe and Denmark (with a final resolution of 5.56 km x 5.56 km). The EMEP MSC-W model has generally been run at 0.1° × 0.1° long/lat resolution in this project (which also gives relatively high spatial resolution over Denmark, ca. 10 km x 10 km), except for the source-receptor runs (described below), which used a spatial resolution of 0.3° × 0.2° long/lat. The EMEP domain for all model runs was the standard CAMS domain (Kuenen et al., 2022).

The main model comparison focuses on 2018 and 2019, with all three models run for these years using both the **noBD** (default setup, no bidirectional) and **BD** (with bidirectional) setups. With the DEHM model, calculations have also been performed with the **noBD** and **BD** setups for the years 2010–2017 and 2020–2021. This is to obtain a time series that can be used for investigating the impact of Nr deposition on sensitive nature areas.

To investigate what happens with the dry deposition of nitrogen when the bidirectional flux is implemented in the models, we have focused on annual spatial distributions of deposition of NH_3 and total Nr. The latter consists of chemically reduced species (denoted NH_x and equal to the sum of NH_3 and ammonium, NH_4^+) and chemically oxidised species (denoted NO_y and consisting of all the nitrogen containing species which also include oxygen). The reduced nitrogen – NH_x – is also interesting to investigate, because it includes the direct reaction products from the emitted NH_3, and we therefore include NH_x data to complement the analysis. Since altered deposition patterns also reflect on the concentration distribution, we also analysed spatial distributions of NH_3 concentrations. The bidirectional flux implies re-emissions of NH_3 that are not accounted for in the anthropogenic emission inventory, and we therefore also examine the magnitude and distribution of bidirectional re-emissions in comparison with anthropogenic emission distributions.

When implementing new parameters or functionalities in the models, it is necessary to evaluate the results through comparison with measurements. This has been done for all the models, based on a database with NH_x observations for Denmark, Sweden, Norway, the Netherlands and France, compiled within the project.

The EMEP MSC-W model was also used to calculate “source-receptor” relations for Nordic countries and closest neighbours with the **noBD** and **BD** setups. In such runs, the NH_3 emissions from a specific country are reduced by 15% and the changes in Nr-deposition across all European grid cells are calculated with respect to a base-case run with no reductions. Note that this 15% reduction is a reduction of the official emissions, not those added through the bidirectional process. As noted above for these runs, a horizontal resolution of 0.3° × 0.2° long/lat was used. Calculations were made for eight emitter countries, selected either because they are Nordic countries or because they were expected to significantly impact the Nordic area.

Finally, to understand the results in more detail, we have investigated the behavior of some of the parameters in the dry deposition routine in the EMEP and MATCH models for selected stations in Denmark, Sweden, Norway and the Netherlands for 2018 and 2019. The results of this investigation are also presented in this report.

Emissions of NH_3 included in the models

Anthropogenic emissions of NH_3 primarily arise from agricultural activities, with a small contribution from other sources, such as traffic (related to catalytic converters in cars) and ocean water processes. Ammonia emissions display a strong annual cycle, because the emissions are tied to fertilisation processes, including the spreading of manure on fields, which in most countries is regulated to only take place when there are growing crops on the fields. Also, the emission of NH_3 depends on the ambient temperature of the air, with increasing emission with increasing temperature (Gyldenkærne et al., 2005; Hertel et al., 2012).

In these simulations, the DEHM model uses data from a detailed national inventory for Denmark, with a spatial resolution of 1 km x 1 km (Plejdrup et al., 2018), combined with the officially reported and gridded EMEP emissions for the rest of Europe (Redeyoff et al., 2024). The MATCH model uses

CAMS-Reg (V5.1c) emission data (Kuenen et al., 2022, Kuenen et al., 2023) and the EMEP MSC-W model uses the official gridded EMEP emissions for the whole domain.

The DEHM model uses standard temporal emission profiles for the various emission sectors based on the profiles from the GENEMIS/EURODELTA projects together with national methods for the temporal distribution of ammonia (Gyldenkærne et al., 2005; Skjøth et al., 2011), whereas the MATCH and EMEP MSC-W models use similar CAMS-Reg-Tempo data to distribute emissions in time (Guevara et al., 2021, Guevara Vilardell, 2023).

Figure 1 shows the annual gridded anthropogenic emissions used by the three models for the simulations for 2018 (the distribution is similar for 2019). Geographical areas with a large fraction of agricultural activity (such as the Netherlands, Northern France, Western Denmark and Northern Italy) are evident with high annual NH_3 emission fluxes. Differences in the features of the emission fields between the models can be attributed to the differences in model grid definition and resolution (DEHM and MATCH apply nested setups, whereas EMEP operates on the same resolution throughout the displayed domain) as well as the source of emission data.

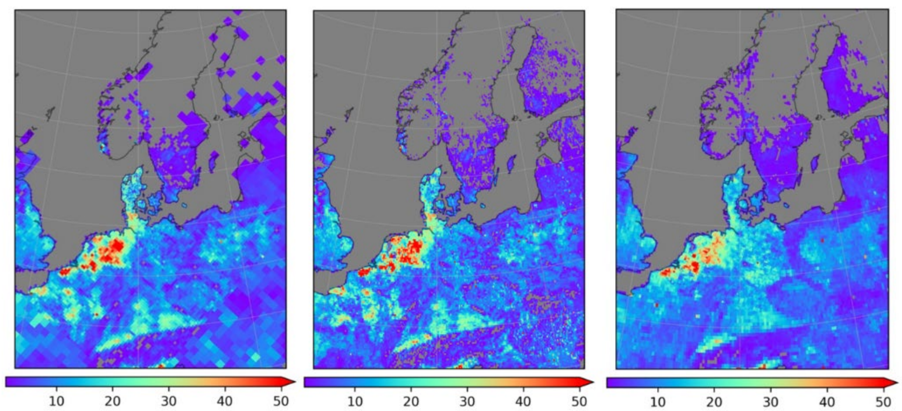

Figure 1: Annual total anthropogenic **emissions** of NH_3 (in kg N/ha) used in the DEHM (left), EMEP (center) and MATCH (right) models for both the **noBD** and **BD** simulations for 2018.

Results

In this section, we focus on NH_3 and present the geographical distribution of concentrations, dry depositions and bidirectional emissions and compare them between the models as well as between the model runs with the **noBD** and **BD** setups, respectively.

Concentration distribution of NH_3

The distribution of the concentration levels of NH_3 will typically reflect the distribution of emissions, and this is seen for all three models, when the results are displayed for a zoom-in over Denmark for 2018 in Figure 2 and 2019 in Figure 3. The figures show the ambient air concentration of NH_3 for the **noBD** and **BD** model runs and the difference between the two concentration distribution fields for all three models.

There are some differences between the models, which reflect differences in model parameterisations, as well as underlying data such as emissions and land cover data. More specifically emissions is one of the driving factors, affecting the overall spatial distribution and magnitude of the NH_3 concentrations among the models. An example is the results from the MATCH model, which shows relatively smaller NH_3 concentrations, corresponding to the smaller emissions, especially in the southern part of the domain (North Germany). The differences between the **BD** and **noBD** model runs are more attributed to differences in land use data and differences in how the parameterisations are implemented in the respective models. Common for the results of all three models is that the concentrations are typically lower in most of the grid cells when the bidirectional flux process is included, as can be seen in the difference plots (right column in the figures) in Figure 2 and 3. All three models also show overall higher concentrations of NH_3 in 2018, compared to 2019. The higher concentrations are partly explained by the very dry summer of 2018, giving higher emissions of NH_3 in the **BD** scenario, and a slightly longer lifetime of the gas in the atmosphere.

In Figure 4, a comparison of concentrations of ammonium (NH_4^+) between the **BD** and **noBD** setup is shown for 2018 and 2019, calculated with the EMEP model. Ammonium is formed secondarily in the atmosphere from reactions between NH_3 and other airborne gases, and is as such only affected by the changes arising from the implementation of the bidirectional flux to a very small extent, as can be seen from the difference plot in the right column in Figure 4. Also, for NH_4^+, there is a difference between the years with a higher concentration of NH_4^+ in 2018 (top row in Figure 4) compared to 2019 (bottom row in Figure 4). This is also a result of the higher NH_3 concentration in 2018.

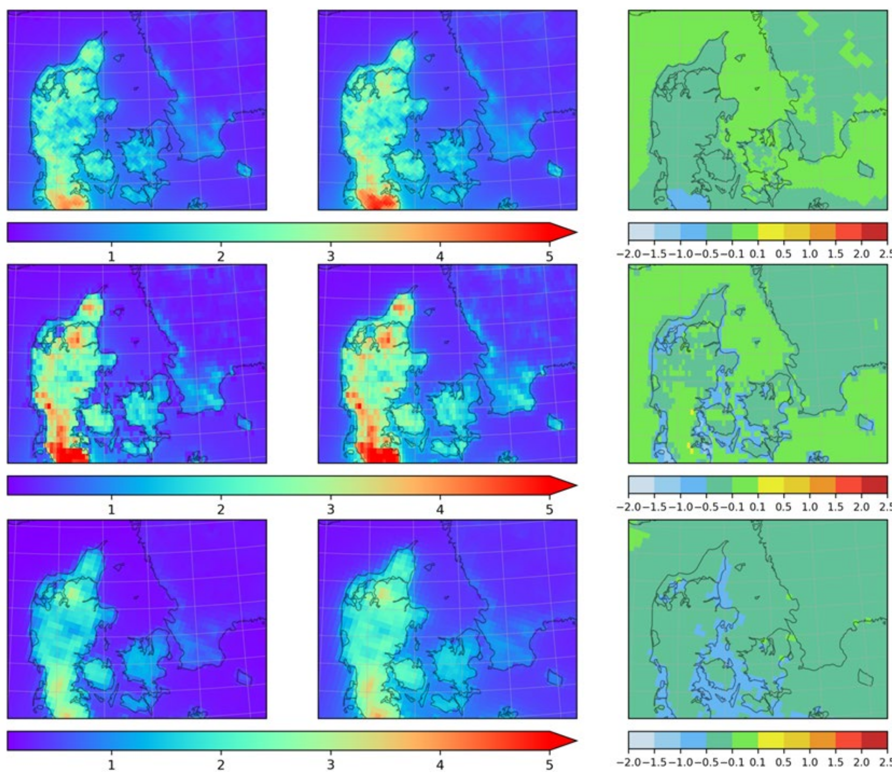

Figure 2: NH_3 concentration maps for 2018 calculated with the models DEHM (upper panel), EMEP (center panel) and MATCH (lower panel). The left column includes bidirectional flux (**BD**), the center column is the original version of the models (**noBD**) and the right column displays the difference between the **BD** and **noBD** model runs (**BD** minus **noBD**). All units are μg/m^3.

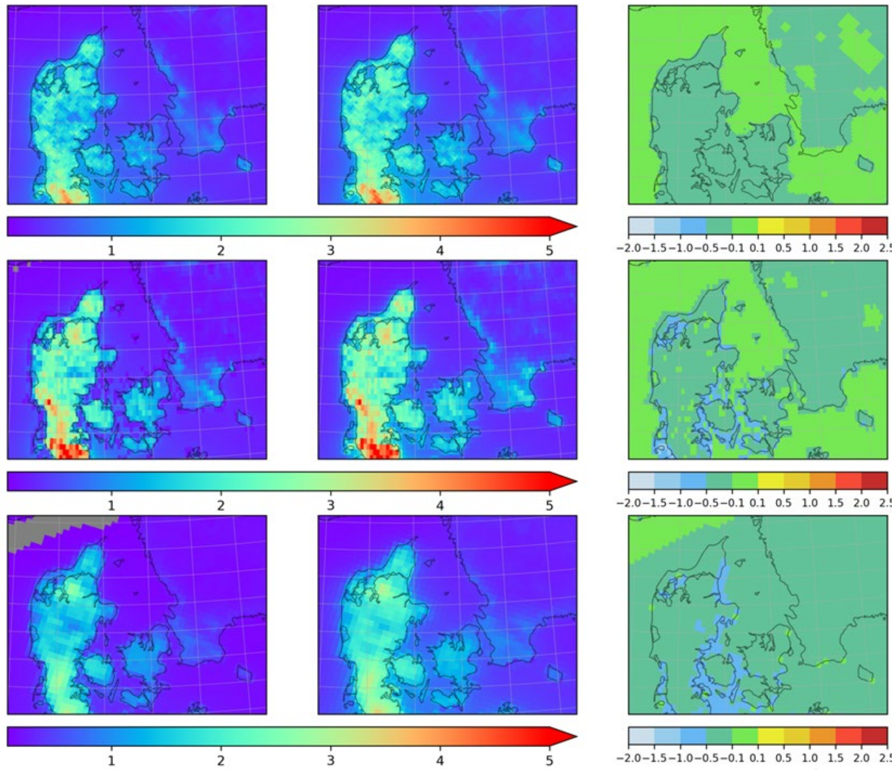

Figure 3: NH_3 concentration maps for 2019 calculated with the models DEHM (upper panel), EMEP (center panel) and MATCH (lower panel). The left column includes bidirectional flux (**BD**), the center column is the original version of the models (**noBD**) and the right column displays the difference between the **BD** and **noBD** model runs (**BD** minus **noBD**). All units are µg/m^3.

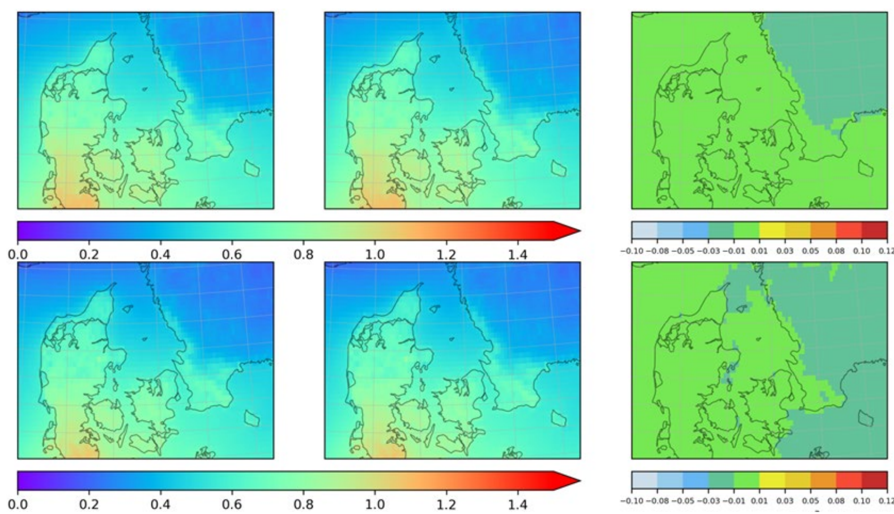

Figure 4: NH_4^+ concentration maps for 2018 (top row) and 2019 (bottom row) calculated with the EMEP model. The left column includes bidirectional flux (**BD**), the center column is the original version of the models (**noBD**) and the right column displays the difference between the **BD** and **noBD** model runs (**BD** minus **noBD**). All units are µg/m^3.

Dry deposition and bidirectional emission distribution of NH_3

Dry deposition of NH_3 occurs relatively rapidly and close to the sources, and therefore the spatial distribution of NH_3 dry deposition also to some extent reflects the emission sources (e.g. Hertel et al., 2012). In the implementation of the bidirectional flux process in the models, the model results are affected in two ways. First, there is an extra emission – the bidirectional emission – and second, the rate of deposition is changed because the parameterisation for calculating the downward flux is different and now depends on vegetation characteristics. In this section, we compare results between the models for total dry deposition of NH_3 for 2018 and 2019 in Figure 5 and 6 (deposition calculated with the **BD** and **noBD** parameterisations), for the net dry deposition (where the bidirectional flux is subtracted from the downward flux) in Figure 7, for the total dry deposition of NH_x in Figure 8 and for the bidirectional emissions of NH_3 in Figure 9.

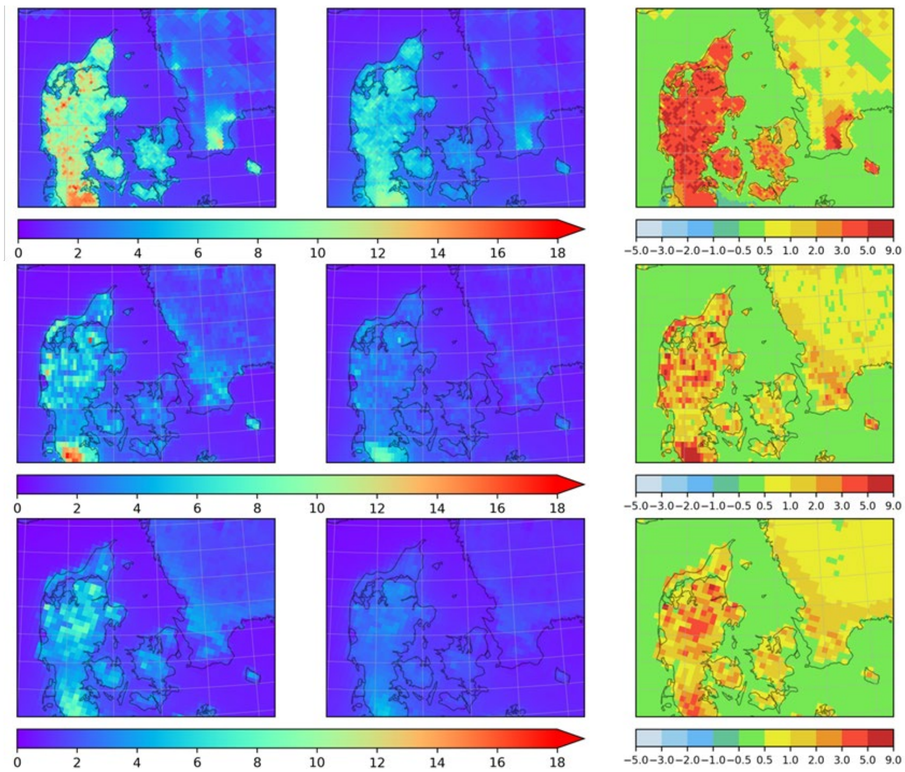

Figure 5: NH_3 total dry deposition maps (in kg/ha) for 2018 calculated with the models DEHM (upper panel), EMEP (center panel) and MATCH (lower panel). The left column includes bidirectional flux process (**BD**), the center column is the original version of the models (**noBD**) and the right column displays the difference in total dry deposition of NH_3 between the **BD** and **noBD** model runs.

Examining the effect of the bidirectional flux implementation on the total dry deposition of NH_3 in Figure 5 and 6, it can be seen that there are some differences between the models in terms of the magnitude of the deposition. Common for all models is that the total dry NH_3 deposition is higher in the **BD** case, so the difference between **BD** and **noBD** is always positive (or close to zero in areas away from NH_3 emission sources, e.g. over the water); this is due to changes in the description of the downward flux in the new parameterisation, where revised surface resistances (r_c) lead to higher dry deposition of NH_3. In addition to differences in numerical coefficients, the new r_c depends on the leaf area index, both for the stomatal depotisoin pathway and for the external leaf pathway. All models show slightly higher depositions in 2018 compared to 2019, but the **BD**-**noBD** difference (right column of figure 5 and 6) is quite similar between the years.

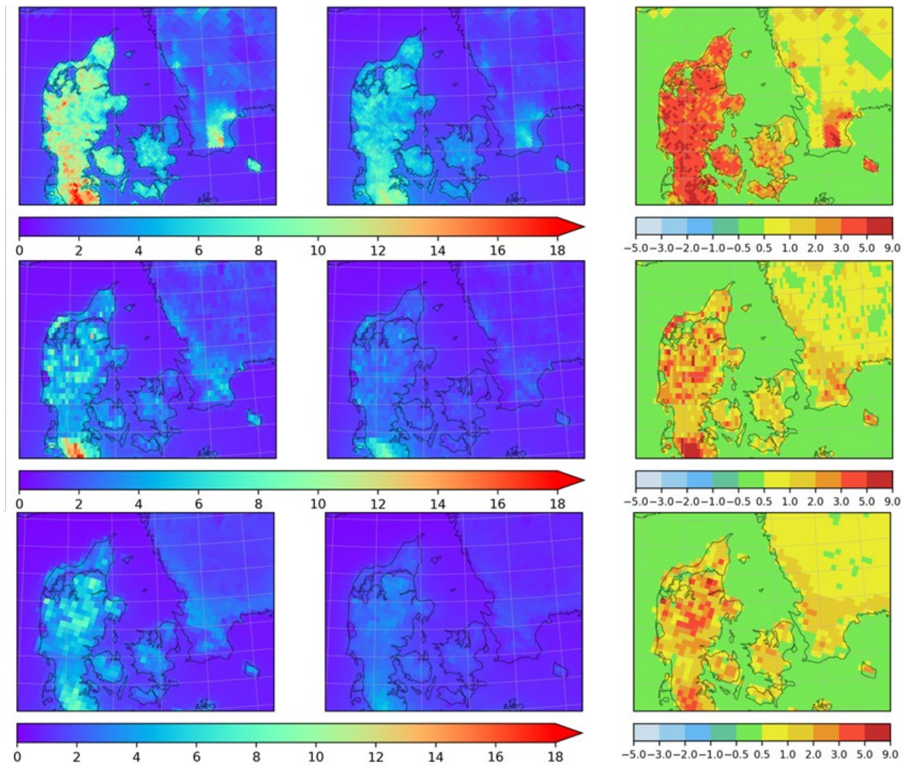

Figure 6: NH_3 total dry deposition maps (in kg/ha) for 2019 calculated with the models DEHM (upper panel), EMEP (center panel) and MATCH (lower panel). The left column includes bidirectional flux process (**BD**), the center column is the original version of the models (**noBD**) and the right column displays the difference in total dry deposition of NH_3 between the **BD** and **noBD** model runs.

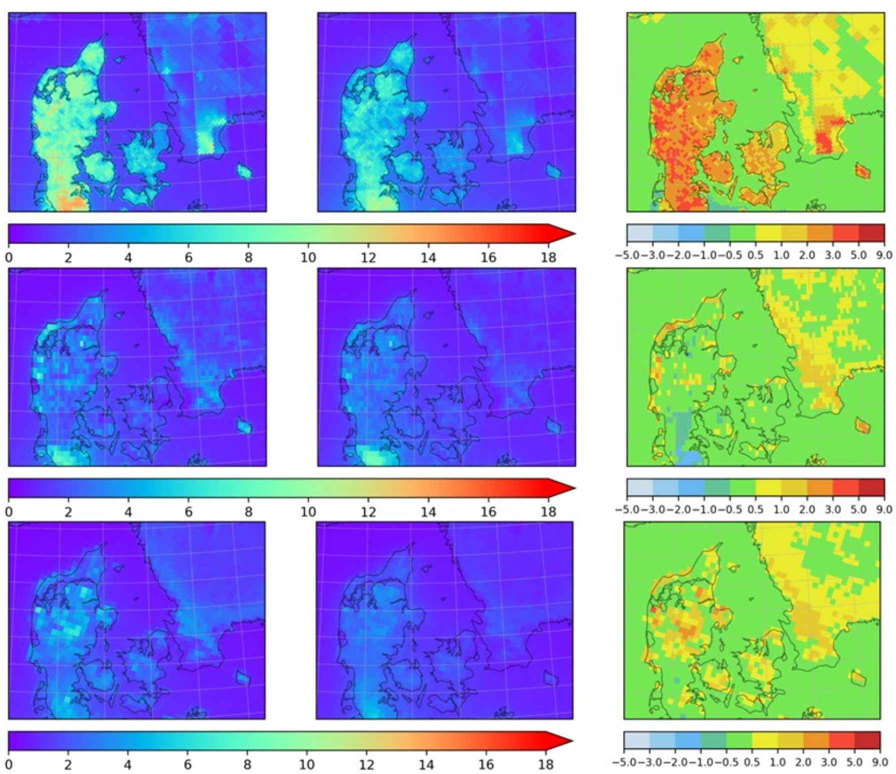

Figure 7: NH_3 net dry deposition maps (in kg N/ha) for 2018 calculated with the models DEHM (upper panel), EMEP (center panel) and MATCH (lower panel). The left column includes bidirectional flux process (**BD**), the center column is the original version of the models (**noBD**) and the right column displays the difference in net dry deposition of NH_3 between the **BD** and **noBD** model runs.

Figure 7 presents the net dry deposition of NH_3, where the bidirectional emission of NH_3 has been subtracted from the total dry deposition of NH_3 for 2018. It is the net deposition of NH_3 that e.g. a forest or other plants at the surface will receive. As was seen for total dry deposition, the results for the EMEP and MATCH models are relatively similar, whereas the net dry deposition in both the **BD** and **noBD** cases is higher in the DEHM results. Also, the difference between the **BD** and the **noBD** case is higher for the DEHM model. This can partly be explained by the high-resolution NH_3 emission data from Danish agricultural activities, used in the inventory for Denmark and the high resolution land use data for Denmark in the DEHM model, but may also be caused by differences in how the parameterisation is implemented. The results for 2019 are similar and not shown here.

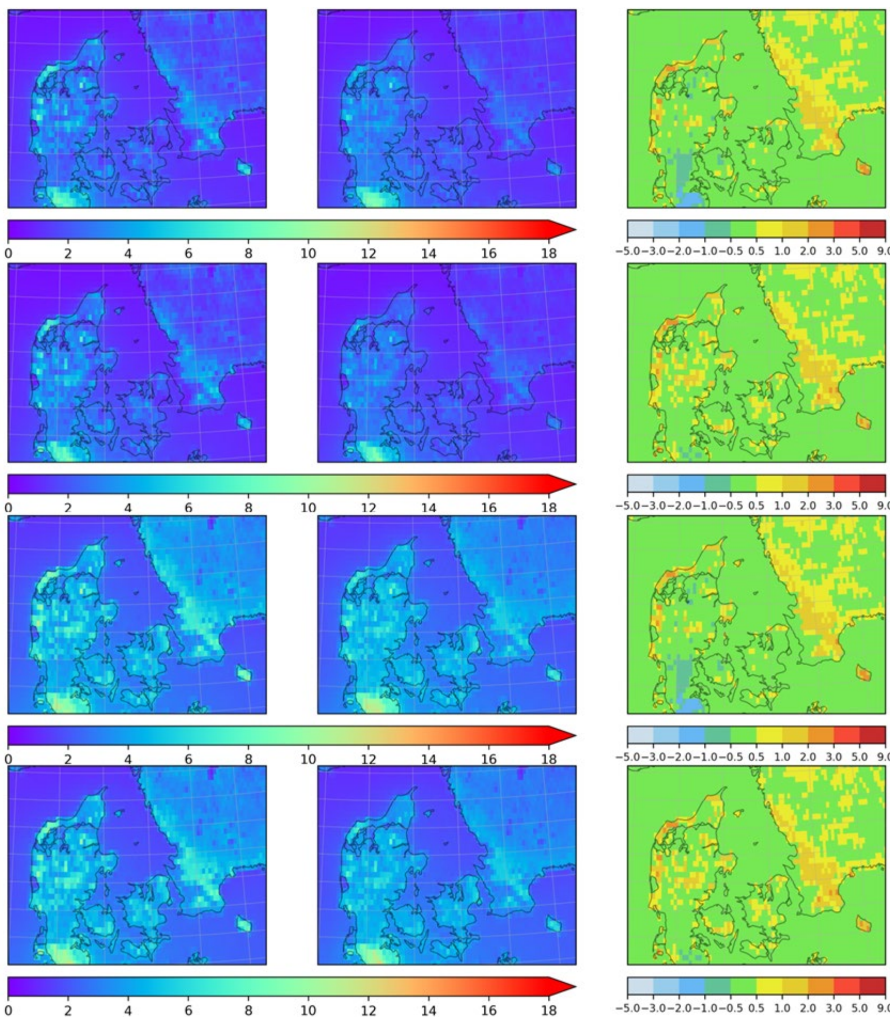

Figure 8: NH_x dry deposition (kg N/ha) for 2018 (1. row) and 2019 (2. row) and Nr dry deposition (kg N/ha) for 2018 (3. row) and 2019 (4. row) calculated with the EMEP model. The left column includes bidirectional flux process (**BD**), the center column is the original version of the models (**noBD**) and the right column displays the difference in net dry deposition of NH_3 between the **BD** and **noBD** model runs.

Finally, in Figure 8, the similar results from **BD**, **noBD** runs and the difference between the two are shown for total dry deposition of NH_x as well as total dry deposition of total Nr. All model runs in

Figure 8 are performed with the EMEP model for 2018 and 2019. The difference between the years for NH_x, respectively Nr, is small. The difference between the **BD** and **noBD** runs is very similar for Nr and NH_x, indicating that NH_x is the primary driver of the variation in the total Nr dry deposition. The extra dry deposition of NH_x resulting from the bidirectional flux implementation, is larger for 2019 than for 2018, and for 2018 even some areas with a slightly negative difference is seen in Southern Jutland/Northern Germany (this pattern is also seen for the total dry deposition, Nr).

The bidirectional re-emission from the three model runs is presented in Figure 9 for 2018. Between 2018 and 2019 (not shown here), there are minor differences, with slightly higher bidirectional emissions in 2018, compared to 2019. When comparing these results with the anthropogenic emissions in Figure 1, it is observed that the largest bidirectional fluxes coincide with the areas that have high agricultural activities, supporting the theory behind the parameterisation. Also, there are some differences between the models, and in this case, they relate to differences in the sources of land cover data as well as the spatial resolution and source of emission data.

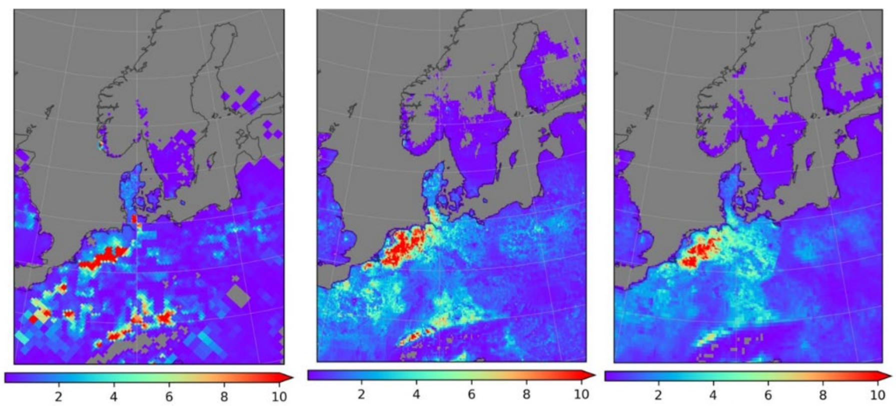

Figure 9: NH_3 bidirectional re-emissions (in kg N/ha) for 2018 resulting from model runs with the DEHM model (left), the EMEP model (center) and the MATCH model (right).

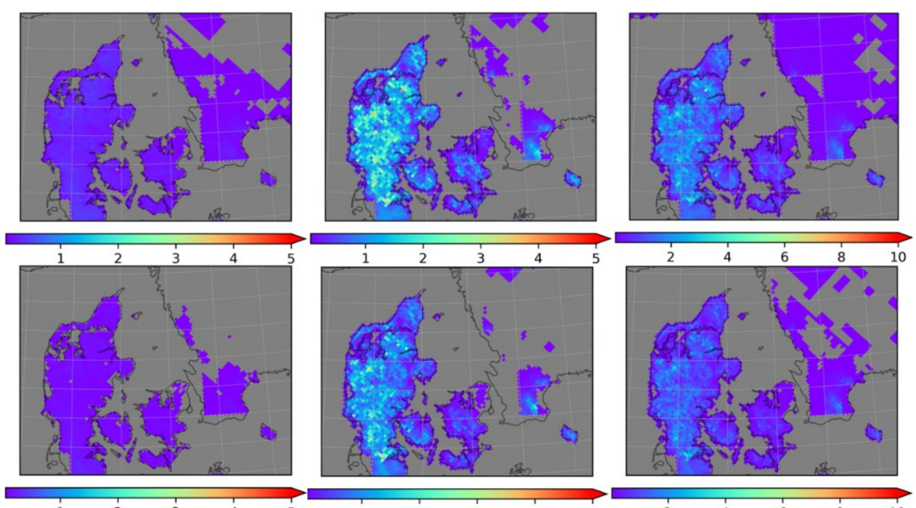

Figure 10: Bidirectional re-emissions (kg N/ha) of NH_3 through the stomatal pathway (left) and the external leaf pathway (center) as well as the total flux (right). All data are based on calculations with the DEHM model for 2018 (top panel) and 2019 (bottom panel). Note the difference in the scales between the plots.

Figure 10 shows the bidirectional flux of NH_3 for the pathway through the stomata and the pathway through the water on leaves (external leaf pathway) as well as the total bidirectional flux, all calculated with the DEHM model for 2018 and 2019. Following the pattern of the deposition, the bidirectional flux is higher in 2018 compared to 2019. When comparing the stomatal and external leaf pathways, it is clear that the bidirectional re-emission is dominated by the contribution from the external leaf pathway in both years, whereas the transport of NH_3 through the plant stomata is of lesser importance.

Source-receptor calculations

The results of the source-receptor calculations are summarised in Figure 11. The emitter countries chosen were Denmark, Finland, France, Germany, the Netherlands, Norway, Sweden and the United Kingdom. Results are presented as country-average changes in NH_x deposition arising from the 15% NH_3 emission reduction in each emitter country. For example, a 15% NH_3 emission reduction from Belgium results in decreases in total NH_x deposition over Belgium of about 0.5 kg N/ha in the **noBD** case and about 1.0 kg N/ha in the **BD** case (Figure 11a). This reduction in Belgium's NH_3 emissions also causes reductions of about 0.1–0.2 kgN/ha in the total NH_x deposition in the Netherlands, but little change elsewhere.

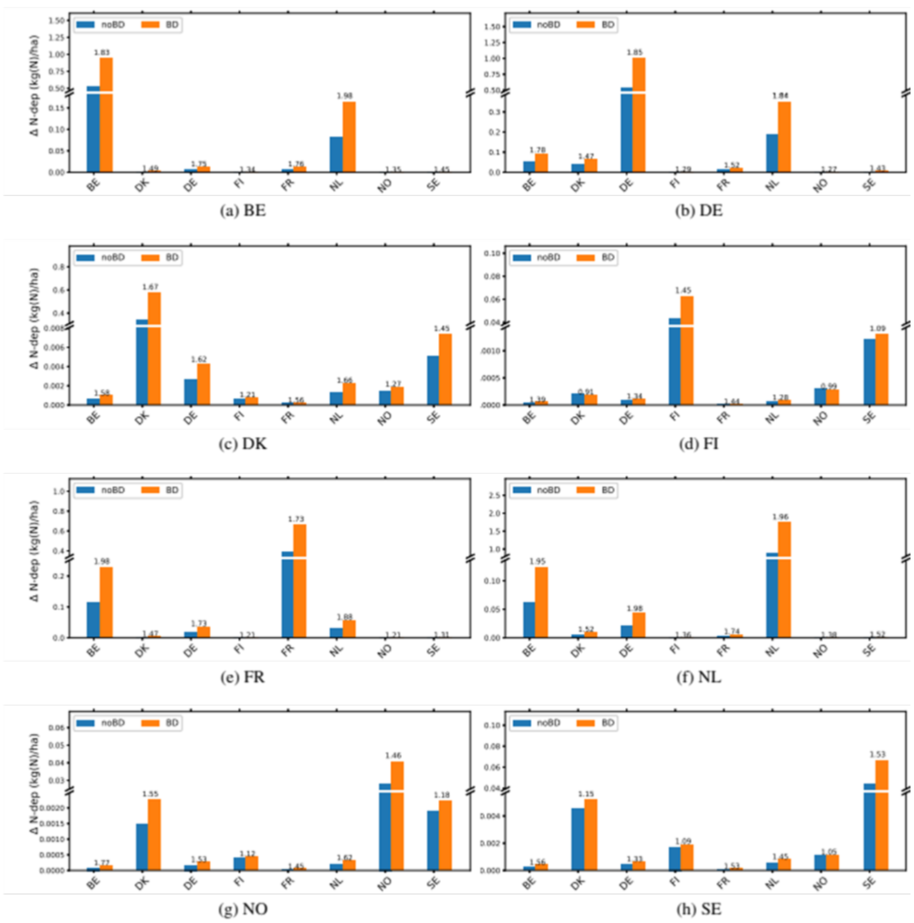

Figure 11: Calculated impact of 15% NH_3 emissions reductions from emitter countries (plots a to h) on total NHx depositions in selected receiver countries (bars for Belgium, BE, Germany, DE, Denmark, DK, Finland, FI, France, FR, the Netherlands, NL, Norway, NO and Sweden, SE). The y-axes shows the country-average change in total NHx deposition. The numbers shown above the bars give the ratio of the **BD**/**noBD** cases. Note the split y-axis, which is

used as most emitters cause substantially bigger changes in their own country than in the other receiver countries. Data are from calculations for 2019 with the EMEP MSC-W model.

It is clear from all plots that the emitters cause substantially bigger changes in their own country than in the other receiver countries (hence the use of the split y-axis). This is expected for any primary pollutant, but in the case of NH_x, an important aspect is that the emitted NH_3 has a high dry deposition, whereas once this component is converted to NH_4^+, the dry deposition rate is much lower; wet deposition is then the major loss process (e.g. Hertel et al., 2012). This means that these components can be transported further away from the emitter country.

It is also clear that the **BD** case produces larger changes in the Nr deposition received by all countries. An interesting feature of Figure 11 though is that the ratio of the **noBD** to **BD** changes (given as numbers above the bars) is not constant. For example, for Danish emission changes (Figure 11c) the ratio ranges from 1.67 in Denmark itself to 1.21 for France, which suggests that the **BD** case promotes more self-deposition than the **noBD** case. However, although it seems that the ratio is usually highest for the emitter country's contribution to itself, and lowest further away, there are exceptions. This complex behaviour almost certainly results from the fact that the **BD** case involves two opposite effects – both extra emissions (which depend on the anthropogenic emission density as well) and increased deposition, as discussed above.

Parameter analysis

In an attempt to understand in more detail what goes on at individual sites, when the bidirectional flux is taken into account, we saved data (time series) of a number of parameters for 11 selected sites in Denmark, Sweden, Norway and The Netherlands (see Table 2). For the results presented here, data were available from the EMEP and MATCH model for the stations Risø, Norunda Stenen and Vredepeel-Vredeweg.

The 11 selected sites were intended to be a mix of examples from nature areas far from sources of NH_3 and measurements closer to source areas. When results from the models were compared to measurements, it turned out that especially the Dutch sites located in the farm areas had very high concentration levels, which the regional scale models did not capture, due to the – with respect to source data – relatively coarse resolution.

Table 2: Stations, locations and characteristics for the 11 selected stations for the parameter analysis in the project.

Station name	Country	Lat.	Lon.	Site type	Notes
Risø	DK	55.695	12.091	Rural	Close to the coast/fjord
Tange	DK	56.354	9.603	Rural	Forest station
Ulborg	DK	56.292	8.428	Rural	Forest station
Råö	SE	57.394	11.914	Rural	Close to the coast
Norunda Stenen	SE	60.086	17.505	Rural	Forest station
Hallahus	SE	56.043	13.148	Rural	Natural site close to DK
Birkenes	NO	58.383	8.250	Rural	Forest, close to farm areas
Vredepeel-Vredeweg	NL	51.541	5.853	Regional	Very close to large farm
De Zilk - Vogelaarsdreef	NL	52.297	4.511	Regional	Nature, close to the coast
Zegweld-Oude Meije	NL	52.138	4.838	Regional	Not entirely nature
Wekerom-Riemterdijk	NL	52.112	5.708	Regional	Close to farm areas

The daily variation of the NH_3 concentration and the deposition velocity for 2019 at the three sites mentioned above are presented in Figure 12. As can be seen from the left part of Figure 12, there are some large differences between the stations, both with respect to the magnitude of the concentration levels modelled, and with respect to the difference between the **BD** and the **noBD** results. For the station in Denmark (Risø, top panel in Figure 12), there is not much difference between the models or the **BD**/**noBD** scenarios for the NH_3 concentration. For the station in Sweden (Norunda Stenen), the concentration level is about half, and the variation both between the MATCH and EMEP models and between the **BD** and **noBD** scenarios is much larger. The Dutch station (Vredepeel-Vredeweg) gives modelled NH_3 concentration levels that are 4-5 times larger than at Risø, and here the variation between days and models is large, but the difference between **BD** and **noBD** is smaller.

The deposition velocity *Vg* is calculated from the resistance to deposition, as described in the Methodology section. When the bidirectional approach is included in the models, it will have an influence on *Vg* through the influence on the surface resistance term. In the right part of Figure 12, the day-to-day variation in *Vg* between the models MATCH and EMEP and the **BD** and **noBD** scenarios is shown, again for the stations Risø (top panel), Norunda Stenen (middle panel) and Vredepeel-Vredeweg (bottom panel). In general, the pattern of *Vg* appears similar between the models and stations, with EMEP (in brown) always showing larger *Vg* values than MATCH (in green). The difference between the **BD** and **noBD** scenarios is quite large for the three stations, especially when *Vg* is high in the **BD** case.

Figure 12: Comparison of the day-to-day variation in NH_3 concentration (left column, unit: parts per billion, ppb) and deposition velocity *Vg* (right column) at the three sites Risø (top panel), Norunda Stenen (middle panel) and Vredepeel-Vredeweg (bottom panel) for 2019. MATCH results are green, EMEP results are brown, **BD** is solid and **noBD** results are dashed. Note the different scales for the y-axis.

Another way to examine the differences between the old and the new parameterization, is to study the dependencies of the non-stomatal surface resistance term (r_{ns}) on parameters such as surface temperature (Ts), relative humidity (RH) and surface area index (SAI). Higher r_{ns} values correspond to higher resistance at the surface to deposition in the parameterisation and therefore gives a smaller deposition velocity and less deposition, if the other resistances are unchanged. Examples of the work with this examination can be found in Figure 13, where the relationship between r_{ns} and *RH*, *Ts* and *SAI* is shown for different setups for the new (**BD**) and old (**noBD**) equations used in the DEHM model. The main difference between the old and the new parameterization is the change in the course of the dependency with concentration level (Figure 13 a, c and d), relative humidity (not shown here) and surface area index (Figure 13 b). Overall, the BD case seems to also demonstrate higher day-to-day variability in both models in Figure 12, which is consistent with Figure 13 showing that the bidirectional flux parameterization is more sensitive to multiple conditions.

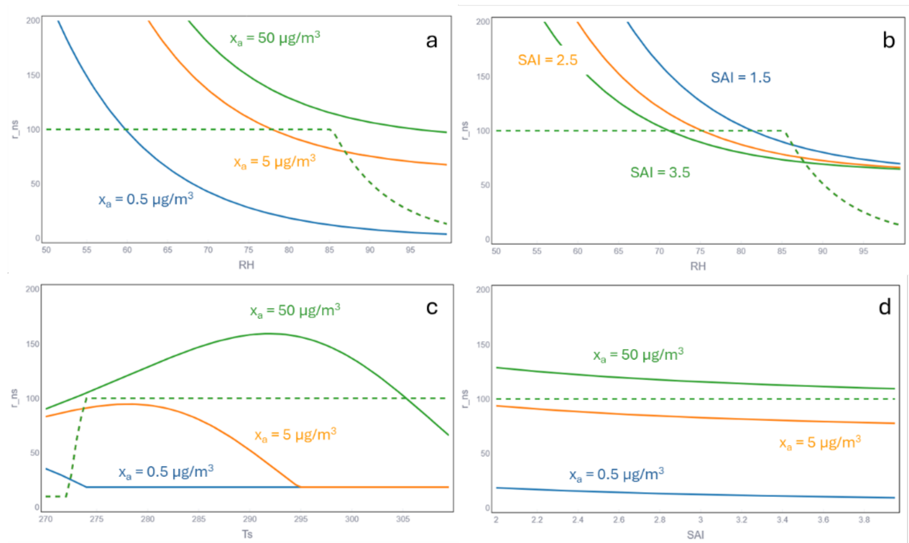

Figure 13: The theoretical relationship between the **BD** case surface resistance r_{ns} (denoted *r_ns* in the plots) and relative humidity *RH* (a and b), between r_{ns} and surface temperature *Ts* (c) and between r_{ns} and surface area index *SAI* (d). In plot a, c and d, the solid graphs correspond to variations of air concentration of NH_3 (x_a). In b the solid graphs correspond to different values of *SAI*. The dashed green line is the original, **noBD**, dependency of r_{ns} in the DEHM model.

If we then also investigate what values the basic variables *Ts* and *RH* have in the two models EMEP and MATCH (Figure 14), then we can see that although the variation is not large, there still are differences, which combined with the alterations given by the change in the basic equations of the dry deposition calculation, will give rise to variation between the model results for the concentration of NH_3 as well as *Vg*.

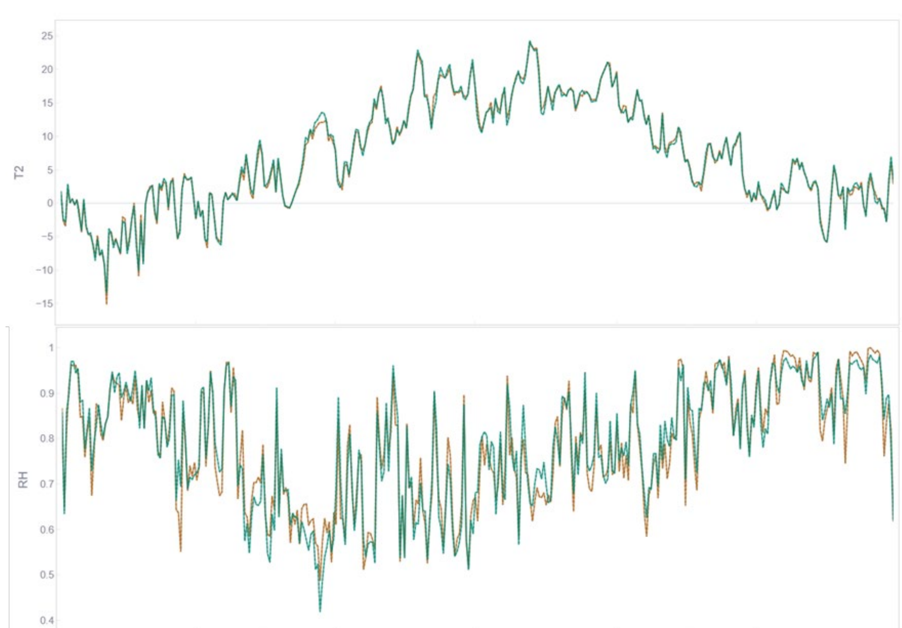

Figure 14: Daily values of surface temperature (Kelvin, top panel) and relative humidity (%, bottom panel) from the MATCH model (green) and the EMEP model (brown) for the station Norunda Stenen in Sweden.

In summary, the parameter analysis has provided insights regarding the changed dependency of key parameters for the dry deposition calculations, both in general and for observational sites with specific conditions, like e.g. high or low ambient concentration levels.

Evaluation of model results with measurements

To evaluate model results with measurements, it would be optimal to have measurements of both atmospheric concentrations and dry depositions of NH_3 and other Nr components to compare with. Unfortunately, dry deposition measurements are extremely scarce due to the complexity of the techniques needed, and not available for the evaluation tasks in this project. We therefore settle for comparison with concentrations of NH_3 and NH_4^+ as the most important components, and NO_2 and SO_4^{2-} for a secondary evaluation.

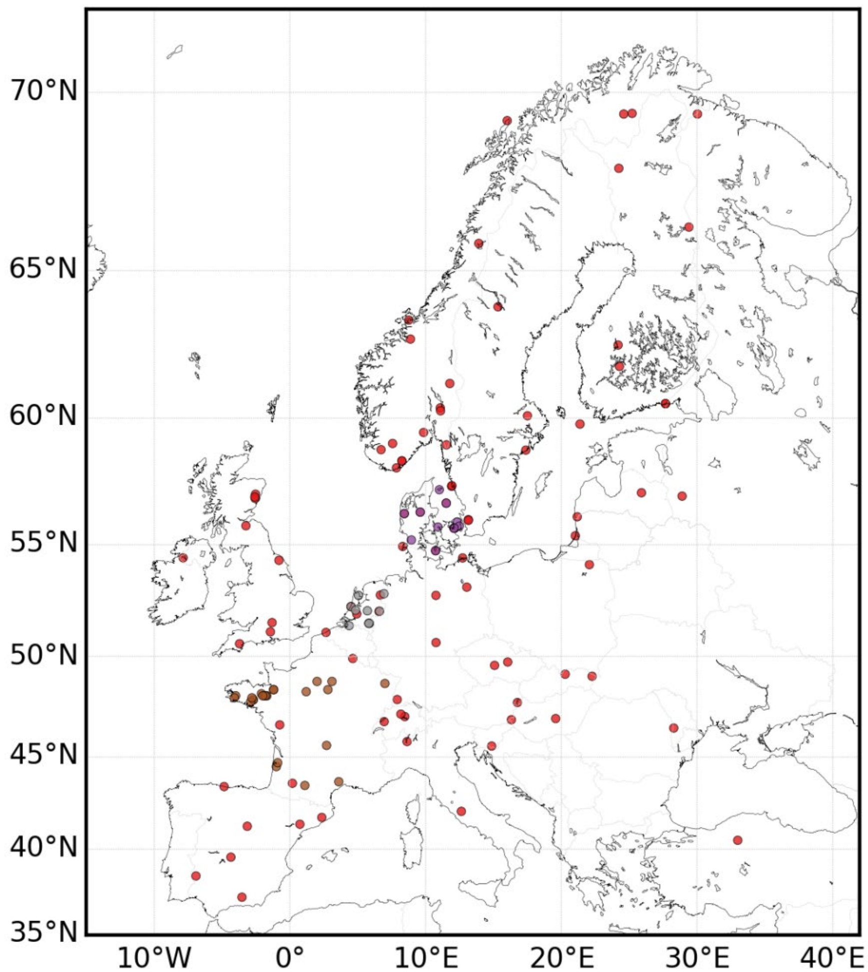

Figure 15: Locations of all NH_x monitoring stations included in the new database. Red dots are stations from the EBAS database, purple dots are stations from Denmark, brown dots are stations from France, and grey dots are the stations from the Netherlands.

EBAS (ebas.nilu.no) is a database with atmospheric measurement data from throughout the globe. In this project, we created a freely available Python package "*pyebas*" (https://pypi.org/project/pyebas/), which provides easy access to the open-source air pollutant data from the EBAS database via their FTP server. The *pyebas* package can both download files from the EBAS database and create a local database for further usage. The downloaded raw EBAS files (in netcdf format) can be exported to .csv files. The locally created *pyebas* database converts ~25GB EBAS raw data to ~800MB of local files. Users can access and query data through the local database.

In this project, we also compiled a new NH_x database with monitoring data from different countries and sources. We collected data from 87 stations from the EBAS database, 11 stations from Denmark (6 of which are also in the EBAS database), 24 stations from the French monitoring network and 10 stations from the Dutch monitoring network. All stations are shown in Figure 15.

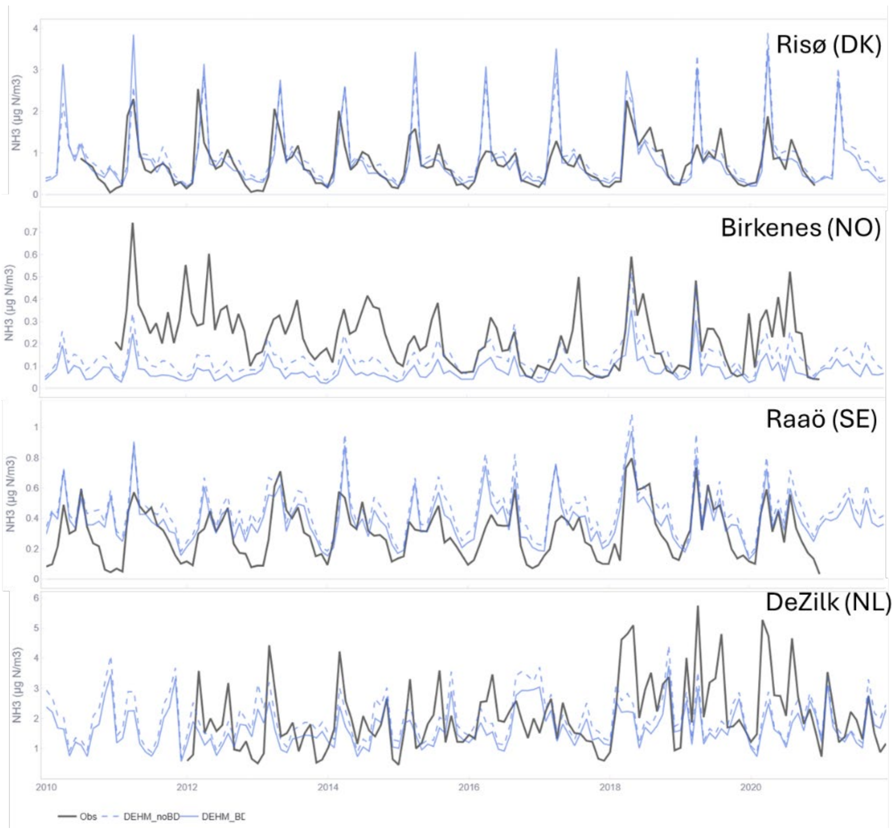

Figure 16: Comparison of NH_3 concentrations at the background monitoring stations Risø in Denmark, Birkenes in Norway, Råö in Sweden and DeZilk in The Netherlands. Results are shown for the time period 2010–2021 as calculated with the DEHM model for **BD** (solid blue line) and **noBD** (dashed blue line) and observations (black line).

To evaluate the model performance and understand the impact of the bidirectional flux implementation, we compared model results to measurements for as many available stations as possible. We have modelled concentrations of relevant components for 2018 and 2019 from all three models, and additionally model results from the DEHM model for the time period 2010 to 2021. Comparisons for single stations are interesting because they can help understand what is going on at individual sites, but comparisons with the mean of all stations are also necessary to evaluate the change in the overall performance of the models as the bidirectional flux process was implemented.

An example of a long-term evaluation for specific stations is shown in Figure 16, where concentrations of NH_3 calculated with DEHM, are compared to observations at four of the rural background stations included in the detailed study (Table 2) for 2010-2021. In this plot, we investigate if the model is capable of reproducing the observed annual and interannual variations. As seen in Figure 16, the model is able to capture the typical variation over the year with a sharp spring peak and a more prolonged, but lower, summer peak for Denmark and Sweden. However, the peaks are typically somewhat overestimated in the spring, and underestimated in the summer. For the Norwegian station (Birkenes), the model consistently underestimates the observed concentrations, although with a better performance for the runs without the bidirectional flux included. For the Dutch station (DeZilk), the model captures the level of NH_3 concentrations, but not the variability over the course of the year. Both the **noBD** (dashed line) and **BD** (solid line) results are presented in Figure 16, and it is seen that the difference between the two setups is not very large in general, but with some differences between the countries.

To evaluate the DEHM model results across the entire model domain, the calculated concentrations of e.g. NH_x have been compared to observations for all available stations in EBAS combined with the Dutch measurements included in the project for the time period 2010–2021. Table 3 presents the statistical results of such a comparison with a focus on the components NH_3, NH_4^+ and the sum of the two (NH_x). The latter is included because, in the observations, it can be difficult to accurately separate the NH_3 gas from the NH_4^+ particles on the filters due to the reversible heterogeneous process that converts NH_3 to NH_4^+. In general, the performance of the model shows high correlation and low bias (see Table 3) with a tendency to show slightly better correlation and slightly worse bias and error, when the results of the **BD** run are compared to the results of the **noBD** run.

Table 3: Results of the statistical analysis of the comparison of observed and modelled concentrations of NH_3, NH_4^+ and NHx for all stations with measurements during the time period analysed. *N* is the total number of observations across stations in the analysis, *Corr* is the Pearson correlation, *FB* is the fractional bias, and *NMSE* is the normalised mean square error. All results are from the DEHM model run for 2010-2021 with (**BD**) and without (**noBD**) the bidirectional flux implemented.

	N	Corr		FB		NMSE	
		BD	noBD	BD	noBD	BD	noBD
NH_3	510	0.86	0.84	-0.41	-0.28	2.5	2.2
NH_4^+	633	0.85	0.84	-0.04	0.01	0.2	0.2
NHx	623	0.79	0.75	-0.09	0.04	1.0	1.0

To evaluate how the DEHM, EMEP and MATCH models perform in comparison for NH_3, Figure 17, top panel shows an example from the station Risø, for the years 2018 and 2019. The spring peak for 2019 is quite shallow, and this is difficult for all the models to capture (especially for DEHM), whereas the spring peak in 2018 is better represented in the models. For the EMEP MSC-W and MATCH models, there is a rather pronounced difference between the **BD** and **noBD** setup of the models in the summer period, with higher concentrations in the **noBD** versions of the models. Examining the residual (Figure 17, bottom panel), it can be seen that the DEHM and MATCH models show a consistent change with lower concentrations in the **BD** case, whereas the picture is more mixed for the EMEP MSC-W model, which for the **BD** case has higher concentrations in the wintertime and lower concentrations in the summertime.

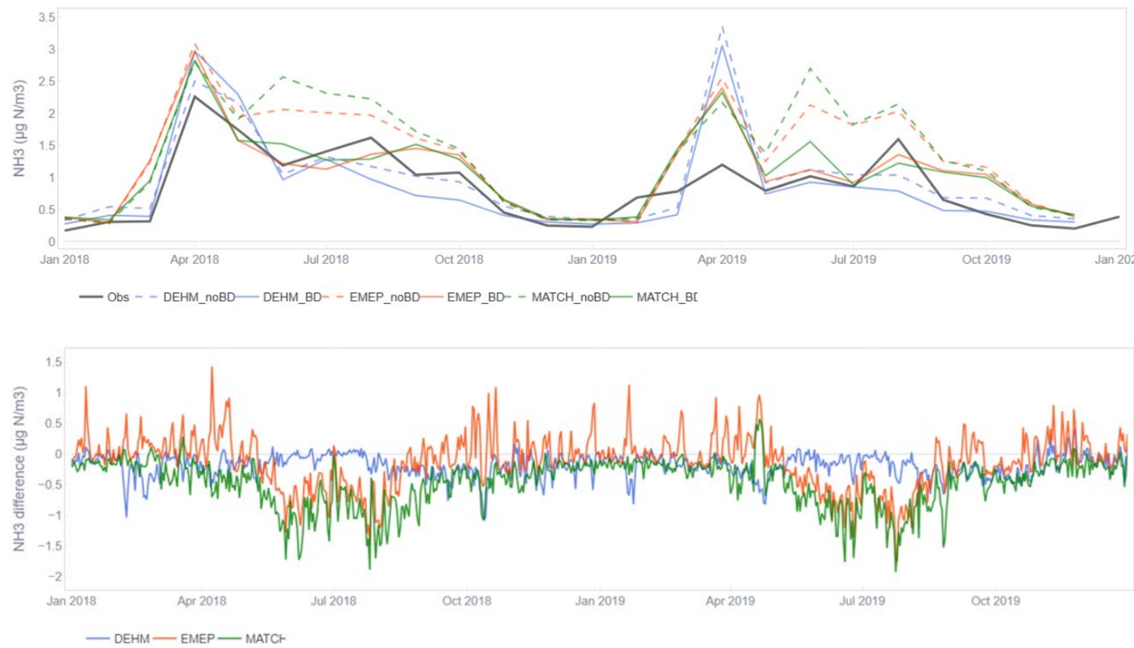

Figure 17: Comparison of NH_3 concentrations (top row) at the Danish background monitoring station Risø, for 2018 and 2019 and the residual between the BD and the noBD runs (bottom row), calculated with the three models DEHM (blue), EMEP (orange) and MATCH (green), and observations (black). Solid lines indicate **BD** model runs while dashed lines correspond to **noBD** calculations.

Changing the NH_3 concentration by including the bidirectional approach potentially influences the formation of the secondary aerosols which contain NH_4^+. Figure 18 presents the same comparison as Figure 17 (top row only), but now for the mean over modelled and observed concentrations of NH_4^+ at the 11 selected stations in Denmark, Sweden, Norway and the Netherlands described in the Table 2. It can be seen that there are very minor differences between **BD** and **noDB** for EMEP and MATCH, and slightly larger differences for DEHM (all drawing DEHM modelled concentrations closer to the observed values).

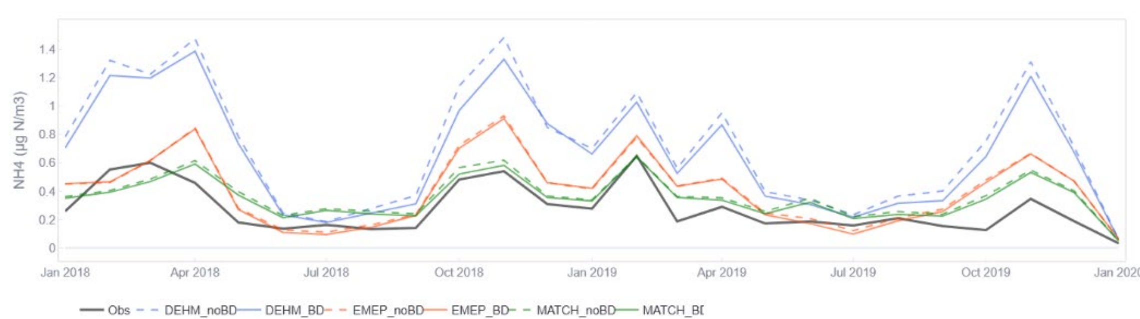

Figure 18: Comparison of NH_{4+} concentrations for the 11 stations described earlier, for 2018 and 2019, calculated with the three models DEHM (blue), EMEP (orange) and MATCH (green), and observations (black). Solid lines indicate **BD** model runs while dashed lines correspond to **noBD** calculations.

Similar comparisons for the gas NO_2 and the secondary aerosol SO_4^{2-} show little or no difference between the **BD** and **noBD** setups (not shown here)

Summary evaluation

The models have been evaluated against measurements of NH_3 from a number of stations. For the three models DEHM, EMEP and MATCH, comparisons are made for 11 stations selected in the case area for the calculations and for 2018 and 2019. To complement this evaluation, comparisons with observations across the DEHM model domain have also been made for the DEHM results for 2010–2021. Overall the **BD** approach has little influence on the concentration of NH_3 when the mean of stations is investigated, whereas for specific stations, larger differences are seen in some cases.

Impacts on Danish terrestrial nature

To understand the impact of including the bidirectional flux process, resulting depositions for Denmark and Danish nature areas are investigated as a case study. In Figure 19, the changes in total Nr deposition from including the bidirectional flux is shown, distributed on land surfaces within the Danish administrative regions and for the country as a whole. It can be seen that the dry deposition of Nr increases by about 30% and the wet deposition of Nr decreases by about 6–7%, giving a combined increase in the modelled total Nr deposition of around 15% between the **BD** and **noBD** scenarios. The changes are smallest for the capital region, but do not differ much across the country.

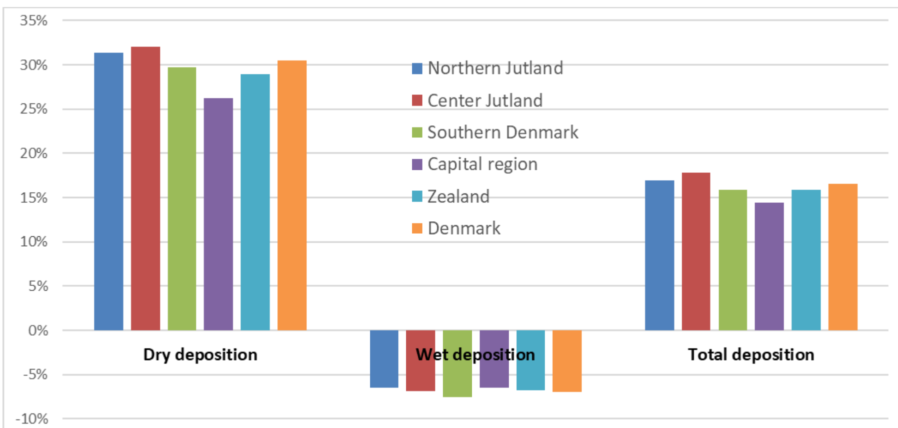

Figure 19: Change in dry, wet and total Nr deposition to all terrestrial surfaces (nature and other) in the five administrative regions in Denmark and Denmark as a whole between the DEHM **BD** and **noBD** model runs.

If we focus on the weighted average of the annual total deposition to specific classes of sensitive nature areas, we can examine the trend over the time period 2010 to 2021 as well as the difference in Nr deposition load between the **BD** and **noBD** approaches. This is presented in Figure 20, where it can be seen, that the Nr deposition trend over the years is not very pronounced (i.e. the deposition level has not changed much on average over this 12-year time period), and that the difference between the two model runs (**BD** and **noBD**) amounts to values of total Nr deposition between ~1 and ~2 kgN/ha for the annual total. For heath, the trends are not statistically significant, but the negative trends of calcereous grassland are significant in both the **BD** and the **noBD** case. Acid grassland is in between with a smaller p-value (<0.05) in the **BD** case.

The trend is important because the response in a nature area to increases, respectively decreases, in Nr deposition can be very different. E.g. the resulting decrease in biodiversity, measured as plant species richness, from increases in Nr deposition typically happens within the following ~5 years, whereas plant species potentially come back to nature areas following decreases in Nr deposition at a much slower pace with some ~30-year delay (Payne et al., 2019).

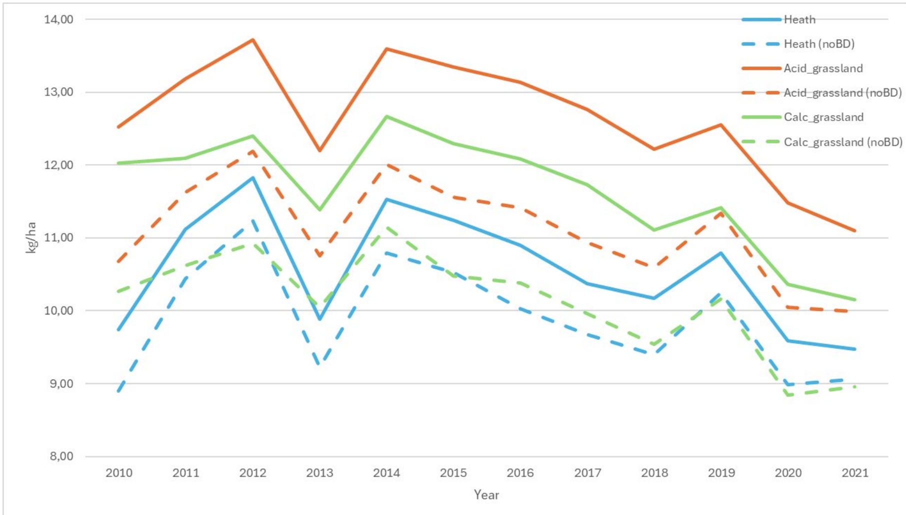

Figure 20: Total Nr deposition as weighted average to Danish sensitive nature areas of the type heathland (blue), acid grassland (red) and calcareous grassland (green). Solid lines indicate **BD** values, dashed lines **noBD** values. Calculated with the DEHM model.

The level of Nr deposition to the sensitive nature areas is important because of the critical load limit that is assigned to the different areas. The critical load range for heathlands is 10–20 kgN/ha pr year, but for detailed impact assessments of heathlands, a value of 10 kgN/ha pr year is more realistic, underpinning the need for accurate deposition calculations (Bobbink et al., 2015).

Conclusions and future work

The reason for including the bidirectional flux process in the atmospheric chemistry-transport models, is to account for the fact that the surface of plants, soil and water are not only sinks of gaseous ammonia. Instead, it is important to consider the concentration of ammonia/ammonium in plants, soil and water, respectively, because if the concentration is higher in e.g. the plant than in the air above, then the ammonia will not be deposited from air to the plant through stomata. Instead, the flux will be reversed. None of the original parameterisations of dry deposition in the three models DEHM, EMEP and MATCH took this process into account, and we thus expected to see a difference from the implementation of the bidirectional flux in the models. However, the intuitive expectation was that because the dry deposition will be modified due to an upward flux from plants/soil/water, the concentrations should increase (because less material is deposited) over source areas and the deposition should correspondingly decrease directly over sources areas, but increase further away from sources. This is because we would expect that plants etc. close to source areas already are saturated with ammonia due to the ongoing emissions, whereas areas further away from sources will have a larger capacity for taking up ammonia through e.g. stomata.

The results presented here – with lower concentrations and higher dry and total depositions in general – are somewhat unexpected, and we have been investigating what is the cause of this, both by analysing results for a subset of stations with high temporal resolution, and by examining the differences in the theoretical relationship between parameters. It appears that the traditional parameterisations of dry deposition (as included in our original chemical-transport models) use surface resistance (r_c) values which, although not accounting directly for the bidirectional flux, already account to some extent for the impacts of bidirectional exchange. In the bidirectional approach of Wichink Kruit et al. (2010; 2012), new r_c values are specified – which depend on surface parameters like e.g. the leaf area index – and these lead to greater losses of ammonia at the surface through deposition, and hence lower concentrations. This increased loss process more than compensates for the extra emissions arising from bidirectional exchange, thereby masking the effect of the altered deposition pattern.

The comparison of the model results with measurements shows that the models perform well for the concentrations of the main nitrogen compounds, but also that there is not really a big difference in performance when the bidirectional flux process is included. Unfortunately, there are no available dry deposition measurements to compare the model results with.

An attempt to understand the impact of the changed deposition was made for Denmark, showing an increase in total deposition (wet and dry) of around 15% for the bidirectional flux scenario. Clearly, more research is needed before the model results of these regional models with the bidirectional flux process included are ready to be used in assessments.

The *Benefit Nature* project

Lise Marie Frohn, Christopher Andersen, Camilla Geels, Jette Bredahl Jacobsen, Thomas Laage-Thomsen, Kristin Magnussen, Julian Massenberg and Bjarni D. Sigurdsson.

Impact assessments of health effects from exposure to ambient air pollution has been under rapid development over the last decades (Ostro & Chestnut, 1998; Pope & Dockery, 2006; Kampa & Castanas, 2008; Manisalidis et al., 2020; Khomenko et al., 2021; Im et al., 2023). Based on systematic reviews and meta-reviews of health effects, it has been possible to derive functions for morbidity- and mortality-related human health responses to air pollution exposure that behave similarly across countries and populations (WHO, 2021). Utilizing the extensive registers available in the Nordic countries, these functions have been investigated in great detail for the Nordic populations (see e.g. Hvidtfeldt et al., 2019; Im et al., 2019; Kukkonen et al., 2020; Lethomäki et al., 2020, Raaschou-Nielsen et al., 2020 and Sommar et al., 2021, which all present results from the Nordforsk funded NordicWelfAir project).

At Aarhus University, we have developed a system for impact-pathway assessments of the consequences of air pollution on human health – the Economic Valuation of Air pollution system (EVA, Brandt et al., 2013a; 2013b; Geels et al., 2015; Anenberg et al., 2015). It is based on high-resolution 3D air pollution exposure data from state-of-the-art air pollution models, recommendations from WHO (2021) with respect to exposure-response functions for human health, and value functions derived from studies of the value of a statistical life, as well as register-based direct cost functions from hospitalization and medication. This system has now been widely applied for policy recommendations in Denmark (e.g. Andersen et al., 2019; Jensen et al., 2024a; 2024b; Nordstrøm et al., 2024) and in the EU (e.g. as an integrated part of the Horizon CL5 project "MARCHES", https://projects.au.dk/marches).

For sensitive nature areas, the situation is much more complicated, as the responses and effects are less systematically studied, not documented for large cohorts, and confounders represent a whole range of processes that occur in natural systems and have a strong geographical, meteorological and temporal component. Acknowledging the inherent complexity of ecosystems, and that environmental impacts of airborne reactive nitrogen (Nr) on ecosystem services in sensitive terrestrial nature are likely to be characterized by a high degree of complexity, the first step of the project was to identify and summarise current methodological challenges and research gaps in the field, thus providing direction for future research and assessment within this area.

The goal of the *Benefit Nature* project, from which results are presented here, was to establish a science base for developing an impact-assessment system that can assess the costs and benefits from effects on biodiversity as a consequence of Nr deposition to sensitive land-based nature areas. The aim is that the system – *EVA-Nature (EVA-N)* - should be based on the same impact-pathway methodology as the EVA system (Friedrich, 1997; Friedrich & Bickel, 2001), following the air pollution from source emissions via atmospheric dispersion and transport, chemical transformation and deposition, to exposure and response from exposure, and ending with the socio-economic valuation of the response. The fundamental methodology of the impact-pathway approach (IPA) is outlined in appendix A.

The research questions of *Benefit Nature* were:

- Is it possible to assess the socio-economic costs and benefits of **terrestrial biodiversity impacts** in a way similar to health impact assessments?
- Are there **exposure-response functions** to be used?
- How can changes in sensitive nature status/biodiversity be **valued**?
    - And what is the **metric** for response and valuation?
- What should be the fundamentals of an EVA-N system following the findings of the above questions?
- Would it be possible to use the EVA-N for answering questions like:
    - What **measures** are the most relevant in the ***agricultural green transition*** to provide the largest **benefits** from the reduction of reactive nitrogen deposition?

Methodology

The project activities were designed to answer the research questions and have included three literature reviews with a focus on valuation functions, exposure-response functions, and impact-pathway approaches for biodiversity response to Nr deposition, respectively. Following the results of the literature reviews, a first version of the EVA-N system has been established.

The first two literature reviews were critical to establish value- and exposure-response-functions for impacts on biodiversity as a consequence of Nr deposition. The third review was not critical for the establishment of a system for assessment of impacts on biodiversity and sensitive nature as such, but served as a discussion frame for the IPA methodology applied in the project.

To apply the IPA to a pollution problem, it was judged necessary that the same, or similar, responses were considered in each of the three literature reviews, thus coupling emissions (e.g. kg NH_3/yr) with costs (e.g. €/kg NH_3/yr) through the processes of atmospheric dispersion, exposure, response, and impact in the endpoint.

The literature reviews were performed in Covidence, a software for managing and streamlining reviews, which allows reviewers to upload, screen, and assess the eligibility of individual papers, and to construct flow diagrams of the results. Covidence is applicable to the PRISMA guidelines (i.e., preferred reporting items for systematic review and meta-analysis; Page et al., 2021) which are designed to conduct and report reviews in a systematic and transparent way. Covidence may be accessed through their website with an account: https://www.covidence.org/.

EVA-Nature system establishment

The focus of establishing EVA-N was determining the best possible and most accurate way of conducting calculations in each of the steps from emissions to socio-economic costs (see Figure A). In 2023, agricultural NH_3 emissions contributed ~67% to Nr deposition in Denmark, while emissions of nitrogen oxides from combustion processes contributed the remaining ~33% (Ellermann et al., 2024). The sources of agricultural emissions are both national and international, with the largest part having national origin. For the combustion sources, the majority of the emissions contributing to Nr deposition in Denmark, originate outside Denmark. The national NH_3 emission sources are very well known for Denmark, due to the comprehensive agricultural registers and the obligation of farmers to report activities with a high degree of detail. In this first version of the EVA-N system, we have relied on regional air pollution modelling only, using the state-of-the-art Danish Eulerian Hemispheric Model – DEHM – which has been developed at Aarhus University for the past 30 years (Christensen, 1997; Frohn et al., 2002; Brandt et al., 2012; Geels et al., 2021; Frohn et al., 2021). In the future development of the EVA-N system, we will also include local-scale modelling of Nr deposition.

In combination with high-resolution land use data from Basemap04 (Levin et al., 2017; Levin et al., 2022), DEHM was used for calculating the Nr deposition to all Danish land areas, including weighted averages to the different nature area categories from Basemap04. These data and results are presented later in this report. One of the effects of Nr deposition on terrestrial nature is the potential alteration of plant community composition and thereby biodiversity. This is the focus in the project here – the natural, or semi-natural habitats.

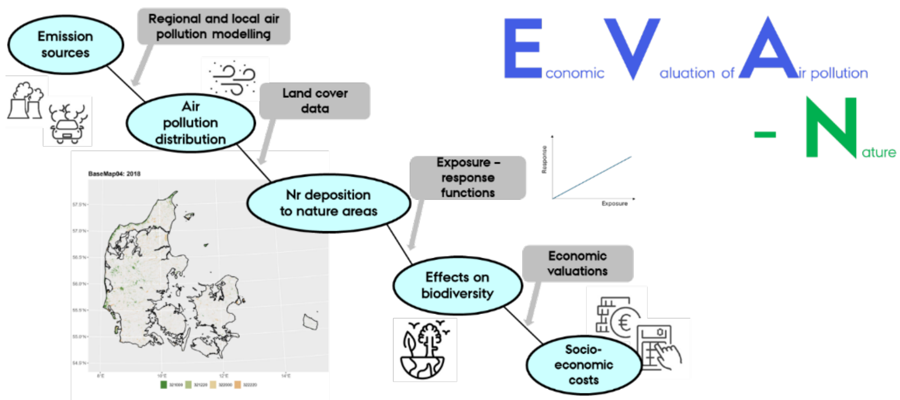

Figure A: A schematic illustration of the EVA-N system describing the impacts along the pathway from emissions, through air pollution distribution and Nr deposition, to effects on biodiversity and corresponding socio-economic costs.

To perform rigorous calculations of effects on biodiversity, many studies of the relationship between exposure and response are needed. For the human health effects of air pollution exposure, there exist large volumes of data, compiled in meta-reviews and analyzed by e.g. WHO (WHO, 2021, and e.g. Amini et al., 2024; Chen et al., 2023).

For changes in biodiversity as the response endpoint, we need to first decide on a specific metric for biodiversity that we can use in EVA-N, then find exposure-response relationships that have sufficient evidence to be used across different sensitive natural ecosystems of the same type. Early in the *Benefit Nature* project, we limited ourselves to focus on ecosystems with low vegetation (e.g. grasslands, heathlands etc.) and analyzing plant species biodiversity only. Since we aim for a system that can be used more generally, without specific knowledge of e.g. exact plant community structure in a given ecosystem, we opted for plant species richness at a given time as the biodiversity metric for calculating response. With this metric, an exposure-response relationship will indicate changes in species richness (decline or increase) but does not deliver insights about what specific species are disappearing (e.g. the most threatened ones) or (re-)appearing, which would be valuable information for designing the value function. However, to be more specific than plant species richness, much more site-specific input information would be required which would make it difficult to use the system in a generic way.

To establish a methodology for calculating the effects on biodiversity from the Nr deposition, and similarly to determine a value function for these effects, search strings were compiled jointly in the project group for conducting the literature reviews. To examine the consequences of Nr deposition to Danish nature areas, time series were established with the DEHM model for 2010 to 2021. Based on the Nr deposition data for this time period, changes in similar plant species richness for the nature types *heath*, *grassland*, and *meadow* were calculated using the exposure-response functions derived from the literature review.

Results

In this section, the results of the three literature reviews are presented first, followed by a summary of the development status of – and the first results from – the EVA-N system.

Review for valuation function

The valuation review aimed to synthesize results from existing valuation studies that address impacts incurred by Nr depositions in sensitive land-based nature types in the global North. By way of adopting the IPA analytical framework for quantifying environmental impacts of a pollutant, we employed the method of a systematic literature review (Page et al., 2021) to identify and derive unit prices from studies valuing environmental impacts, preferably exemplified by damage to biodiversity in terms of losses in similar species richness (SSP). Ideally, these unit prices could then be incorporated into the EVA-N system by using the value transfer methodology (also known as the benefit transfer method). If successful, the EVA-N system can be used to evaluate the effects of interventions that regulate atmospheric emissions of nitrogen, by estimating the socio-economic benefits for nature (and biodiversity) from reductions in Nr depositions.

This literature review has several relevant interfaces with two existing literature reviews that aimed to couple the impacts of nitrogen deposition and resulting societal costs (see Magnussen et al., 2021 and Olsen et al., 2021). Since these studies employed research inquiries similar to those, we specify in the *Benefit Nature* project, we first present and discuss their overall approaches and findings within the context the project. Second, building on these insights we refine the research inquiry and eligibility criteria employed in this review. Third, we summarize the key findings of the review and highlight the current lack of relevant valuation studies.

Finally, we propose a framework for including the costs of biodiversity loss from air pollution, drawing on the green net national income approach developed by Jacobsen and Lundhede (2024).

Background

Magnussen et al. (2021) sought – on behalf of the Norwegian Environmental Agency – to update the existing knowledge base for valuing the environmental impacts of various airborne pollutants in Norway. The primary research objective in Magnussen et al. (2021) was to assess environmental costs, preferably estimated as unit prices, incurred by the detrimental effects of airborne pollution on ecosystems, habitats, and species (i.e. €/kg emission of ammonia (NH_3) nitrogen oxides (NO_x), sulphur dioxide (SO_2) and ozone (O_3)). The rationale for this is that, to make the socio-economic impacts of air pollution evident, unit prices are needed so that the effects on human health and the environment can be quantified and monetized in subsequent socio-economic analyses (i.e. Cost-Benefit analysis).

Olsen et al. (2021) reviewed the current knowledge base and methods for quantifying and monetizing environmental impacts of changes in the atmospheric depositions of NH_3 in Danish nature. On this basis, they discuss and synthesize results (i.e. unit prices) from existing Danish valuation studies based on citizens' preferences (i.e. primary valuation), shadow prices and restoration cost, respectively. As such, Olsen et al. (2021) explored the potential for coupling unit prices with environmental impacts of NH_3 emissions on non-forested terrestrial nature types, such as grasslands and heathlands. The primary objective, therefore, was to enable an estimation of the socio-economic benefits of reducing atmospheric NH_3 deposition and to provide guidance for environmental policy to regulate Nr-emissions.

Operating with similar research scopes, both Magnussen et al. (2021) and Olsen et al. (2021) suggested that the current main methodological and informational challenge for reliable value transfers (i.e. benefit transfers) for assessing socio-economic costs related to environmental impacts incurred by Nr deposition (and air pollution in general), requires new (primary) valuation studies.

Experiences from Magnussen et al. (2021)

Much like the research inquiry scrutinised in this literature review, the purpose in Magnussen et al. (2021) was to assess whether there is a sufficient knowledge base for monetizing environmental impacts caused by air pollution from various emission sources.

Magnussen et al. (2021) include both Norwegian and international scientific and "grey" literature (i.e., scientific articles, manuals and guides, reports, books, etc.) but takes a strictly Norwegian perspective. It aims at identifying and evaluating valuation studies from which unit values/costs for air pollution's damage to the environment (ecosystems, species, and habitats) are applicable to a Norwegian setting. After initial searches in several citation databases (including Google Scholar and SCOPUS) and screening of titles and abstracts, 92 studies were selected and included in the actual review. Of these, 21 studies were perceived as having «high relevance» regarding methods and results. Literature of «high relevance» was determined based on the following set of conditions, i.e. eligibility criteria (translated from Norwegian):

A) The study must value environmental impacts in terms of unit prices for relevant pollution components (e.g., €/kg emission of NO_x).
    a. Value estimates/unit prices for human health are not relevant.
B) The study should define environmental effects (e.g., eutrophication, acidification etc.), and value either;
    a. Effects of (deposited) atmospheric nitrogen (emissions of NO_x and NH_3) on terrestrial nature and freshwater nature,
    b. Acidification (emission of SO_2, NO_x and NH_3) on freshwater nature or
    c. Vegetation damage from ground-level O_3.
C) Value estimates should ideally be transferable to Norwegian conditions in terms of:
    a. Relevant pollutants (e.g., airborne NH_3, NO_x, SO_2, or O_3) and
    b. Relevant habitats/species/ecosystems (also found in Norway).
D) Valuation methodology should ideally be based on an IPA or similar.

For a more complete account of the execution, reporting as well as the method employed, see chapter 2 in Magnussen et al. (2021). Despite producing a comprehensive review of the literature, the key findings from Magnussen et al. (2021) suggest that there are few new original primary valuation studies available on the topic from the last 10-15 years. Moreover, newer studies are often based on the findings of older studies, using simplified value transfers (i.e., unit benefit transfer), presenting crude approximations of unit values for the EU or Europe. Only for some countries (i.e. UK, Germany, and Sweden) more specific (and adjusted) value estimates are approximated. As highlighted by Johnston et al. (2021) and Navrud & Ready (2006), temporal and spatial transfer errors often arise when value estimates are transferred between policy and study site without proper use of the value transfer methodology (i.e., adjusted to the policy site).

The Magnussen et al. (2021) review also highlights that relatively little emphasis has been placed on valuing the environmental impacts of air pollution, while considerably more emphasis has been placed on the valuation of health impacts. This may partly be due to a weaker scientific evidence base for the quantification of environmental impacts, including a general lack of exposure data and well-defined exposure-response function relationships. Moreover, since human health-related damage costs are easier to quantify, they therefore (as a result or otherwise) also dominate valuation in existing primary studies. Although value estimates do exist for certain environmental effects, such as the damaging impacts of eutrophication on nature and biodiversity, the effects of airborne pollution are generally neither well-addressed, quantified nor valued.

For assessing damages incurred by airborne Nr deposition on biodiversity, the criterion *exceeding the range (or limits) of tolerance* is most frequently used. The range of tolerance outlines the constraints for successful growth, development, and reproduction in organisms, and exceeding the range leads to organism death. That is, while nutrients such as nitrogen subsidize biological activity at low concentrations, they reduce it at high concentrations. To explore such relationships, however, specific exposure-response functions are needed to measure the range of tolerance and thus estimate the impacts on e.g. species, species-diversity and biodiversity. Environmental impact estimates may subsequently be coupled with transferred value estimates (i.e. unit prices) - which according to Magnussen et al. (2021) typically is done from either preference-based studies or restoration-cost studies - to determine the socio-economic impacts of air pollution. In many cases, however, these value estimates have been transferred in time and space without thorough assessments of their transferability (Magnussen et al. 2021).

In most of the perceived «high relevance» studies, an IPA or Damage Function Approach (DFA) (or similar) has been applied. Although Magnussen et al. (2021) emphasize that such approaches are needed and should ideally be used to establish a consistent methodological standard for valuing the damage impacts of air pollution, they also find that most studies employ simplified versions of the full IPA. This may in part be due to a lack of exposure data and environmental impact documentation and partly due to the absence of relevant value estimates, leading to crude value transfers. Accordingly, there are very few detailed applications of the full IPA in the literature. In this regard, Magnussen et al. (2021) undersores the urgent need for new primary valuation studies that are compliant with the IPA method, i.e. made with value transferability in mind. This is a prerequisite for obtaining more precise and rigorous unit prices related to the specific environmental impacts of eutrophication on terrestrial and freshwater (limnic) biodiversity alike, incurred by airborne pollutants. Despite recognizing the existing information barriers of applying the full IPA to this pollution problem, Magnussen et al. (2021) recommend that an IPA is to be used when evaluating the impacts of air pollution to establish a consistent methodological standard for integrated environmental assessments.

Experiences from Olsen et al. (2021)

Olsen et al. (2021) conducted a literature review pertaining to the valuation of environmental impacts from NH_3 deposition on terrestrial nature types (excluding forests) in Denmark. To address the socio-economic consequences of impacts of NH_3-deposition, Olsen et al (2021) evaluate various methods for quantification (i.e., coupling Nr emissions with environmental impacts) and valuation (i.e., monetizing environmental impacts), respectively, whilst also reviewing the existing knowledge base in Denmark for the quantification and valuation of NH_3 emissions.

Regarding the latter, Olsen et al. (2021) assess the historical developments of NH_3 depositions and the related environmental impacts in Denmark, highlighting the possibilities and limitations for coupling those impacts to specific unit prices from Danish valuation studies. Although NH_3 depositions peaked around the 1990s and have since followed a decreasing or status quo trend (Ellermann et al., 2024), the adverse effects of NH_3 depositions, exemplified by decreasing C/N ratios, are still present in Danish nature areas. Unit prices were derived from three overall categories of valuation studies, including i) damage cost approaches (i.e. preference-based studies), ii) avoidance cost approaches (i.e. based on shadow prices), or iii) replacement cost approaches (i.e. costs of replacing/repairing/restoring the adverse impacts caused by the externality).

Olsen et al. (2021) assess how the effects of NH_3 depositions are quantified using approaches such as critical limits, dose-response relationships, and direct effects. On this basis they evaluate the pros and cons of applying unit prices from the different approaches to valuation (i.e. the three valuation approaches described above). Whilst establishing a negative correlation between species richness (i.e. mean number of species for five 2 m x 2 m quadrants) and total inorganic Nr deposition (kg N/ha/yr.), Olsen et al. (2021) also emphasize that it is indeed only preference-based valuation studies, which are consistent with the welfare economic theory, and therefore is considered to be the preferred (recommended) approach. Using the value transfer approach, value estimates obtained from the study site (i.e. where the original valuation study is conducted) can be applied to a policy site (i.e. where the valuation estimate is needed), provided that such a transfer is carried out according to e.g. Johnston et al. (2021)'s *best practice* recommendations for using the value transfer method for socio-economic valuation of environmental goods. A prerequisite for using the value transfer method is that the study and policy areas need to be as similar as possible in terms of physical conditions, the characteristics (demographics) of the population, and the environmental endpoint for which monetary values are desired. The less similar the areas, the greater the uncertainty surrounding the value transfer.

Like Magnussen et al. (2021), the Olsen et al (2021) study applies an analytical framework whereby the impacts of deposited NH_3 (on Danish grasslands) are quantified and ultimately sought to be valued through a sequence of principal steps, quite similar to those employed in an IPA (see Figure A.1 in appendix A). Although their framework consists of six steps, the core four steps are consistent with an IPA: 1) Source/emission, 2) Dispersion/concentration at receptor sites, 3) exposure-response functions/impacts, and 4) monetary valuation/costs and benefits. The framework developed by Olsen et al. (2021) is depicted in Figure B, which also provides examples of the knowledge requirements needed to reach the next step in the framework. The two remaining (i.e. the first and last) steps suggest that new environmental policies may regulate emissions at the source, and that a successful application of the IPA (i.e. coupling emissions with valuation of impacts in the environmental endpoint) may be used in socio-economic analysis (i.e. cost-benefit analysis) to guide regulation, respectively.

Although Olsen et al. (2021) established an inverse relationship between NH_3-emissions and species richness in Danish grasslands, a complete environmental impact assessment of NH_3 depositions has not been accomplished. As such, the main finding was that it is not readily feasible to make a direct (i.e. an ideal) link between emissions and effects.

Quantifying the effects of nitrogen pollution on nature is complicated by the fact that i) there is both a significant local NH_3 deposition and a contribution of NH_3 to other Nr-containing air pollution components that can be transported over long distances in the atmosphere with the wind, ii) there may be significant time delays between emissions and effects, because a significant portion of the effect will be a result of accumulated depositions (over time), and iii) since the exact impact of exposure differs with regard to nature type and soil management regime.

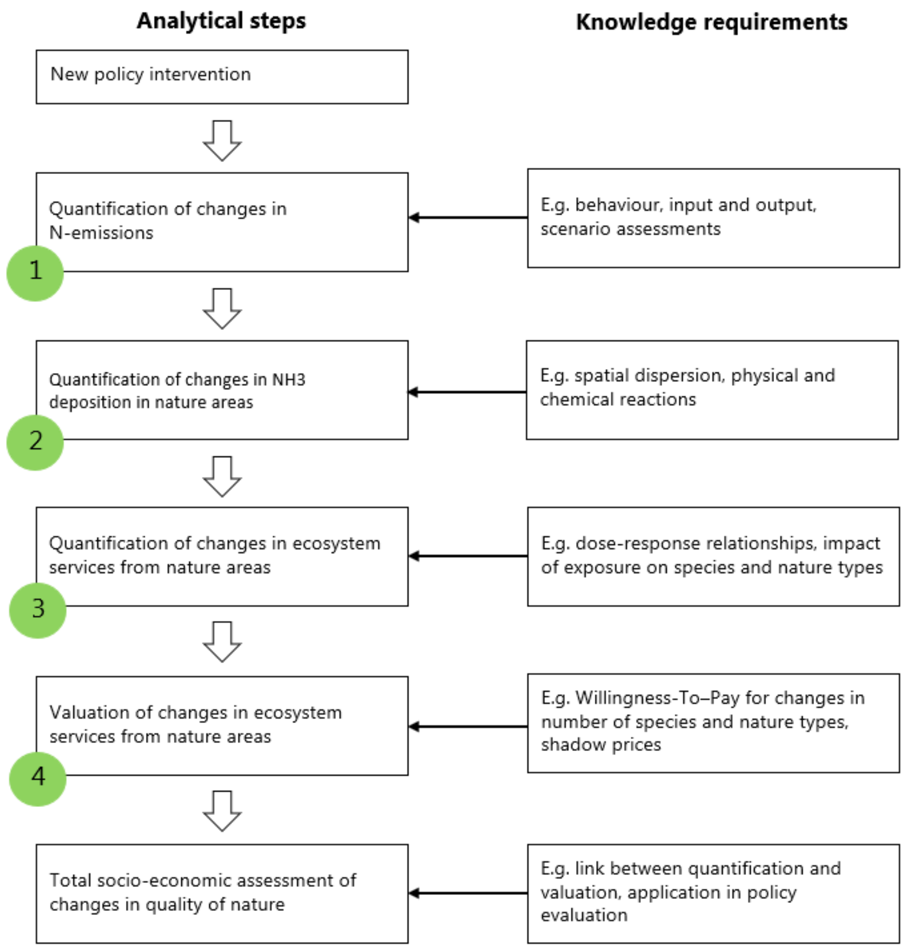

Figure B: Overview of the analytical steps in the socio-economic impact assessment and the related knowledge requirements in Olsen et al. (2021). Numbers (on the left) show the steps analogous to the principal four steps of an IPA, as depicted in Figure A1 in appendix A. *Source: Translated to English from Olsen et al. (2021, p. 4)*.

A total of 12 primary valuation studies were reviewed, from which value estimates (in willingness-to-pay (WTP)/household/yr) for different impacts such as protected areas, species protection, and avoiding environmental bads (i.e. a negative externality) were presented. Olsen et al. (2021) found that it was not possible to make the necessary link between quantification and valuation of impacts on the basis of the existing knowledge base, without making several assumptions, which from a scientific point of view, will be questionable. As such, none of the reviewed valuation studies had been carried out specifically in relation to valuing the impacts of NH_3 deposition on Danish nature. This prevented a coupling of emissions of NH_3 (kg NH_3/yr) with socio-economic costs (e.g. €/kg NH_3/yr) through the processes of exposure to response, since none of the value estimates were quantified in view of the pollution problem per se, thus specified in units applicable to the biophysical quantifications of NH_3 deposition. A main result from Olsen et al. (2021) is that research fields have been developed independently of each other, and consequently, it was not possible to apply an IPA-like method to this pollution problem. This highlights the inherent interdisciplinarity of the problem and underscores the need for collaborations like *Benefit Nature* that bring together researchers from different scientific fields assigned with tasking the different steps along the impact pathway.

According to Olsen et al. (2021), future research efforts should therefore be aimed at establishing the necessary link between quantification and valuation of the environmental impacts of NH_3 deposition. More specifically, that valuation should be related to the same units as the quantified effect and impacts, e.g. in terms of i) area – change in quantity (ha grasslands) or quality (ha grassland in good/pristine status), ii) species – change in quantity (number of species) or quality (status for indicator species), or iii) ecosystem services – change in recreational opportunities or change in carbon sequestration. To this end, Olsen et al. (2021) emphasized a coordinated research effort that has an application to socio-economic impact analysis in mind, to support future policies and regulation of NH_3 emissions.

Search strategy and eligibility criteria

The main research inquiry of the valuation review in *Benefit Nature* was developed and narrowed down on the basis of the context analysis described above and has been conducted in accordance with the overall PRISMA guidelines (see Page et al., 2021). The review should ideally identify all relevant studies that value losses in biodiversity, when sensitive terrestrial nature types in Northern Europe are exposed to airborne Nr pollution, thus coupling adverse environmental impacts with unit prices (€/kg emissions). To this end, a series of search words and strings have been developed.

The search strategy is presented in Table A and B. Searches were performed in SCOPUS and Web of Science (see Table A and B) and produced 1,578 and 698 results, respectively.

Table A. Valuation review, SCOPUS Search

Search string	Search word
Nature type(s)	TITLE-ABS-KEY("natur*" OR "land-base*" OR "terrestrial*" OR "grassland*" OR "forest*" OR "heath*" OR "peat*" OR "sparsely vegetated*" OR "tundra*" OR "mire*" OR "fen*" OR "meadow*")
AND	
Pollution media/ effect/ component	TITLE-ABS-KEY ("air" OR "air pollution" OR "airborne pollution" OR "atmospheric" OR "deposi*" OR "nitrogen" OR "Nr" OR "reactive" OR "nitrous*" OR "NOX" OR "NO2" OR "ammonia" OR "NH3" OR "nitrate" OR "NO3" OR "fertilization" OR "long-term fertilization" OR "nutrient manipulation")
AND	
Valuation component	TITLE-ABS-KEY("valuation*" OR "contingent valuation*" OR "choice experiment*" OR "non-market" OR "hedonic price*" OR "travel cost" OR "avoidance cost*" OR "replacement cost*" OR "WTP" OR "WTA" OR "socio-economic" OR "impact pathway*")
AND	
Value estimation	ALL("cost*" OR "benefit*" OR "unitprice*" OR "externalit*")

Note: This search gave 1,578 results. Date of search: 12/12/2023

Table B. Valuation review, WoS Search

Search string	Search word
Nature type(s)	TS=("natur*" OR "land-base*" OR "terrestrial*" OR "grassland*" OR "forest*" OR "heath*" OR "peat*" OR "sparsely vegetated*" OR "tundra*" OR "mire*" OR "fen*" OR "meadow*")
AND	
Pollution media/ effect/ component	TS=("air" OR "air pollution" OR "airborne pollution" OR "atmospheric" OR "deposi*" OR "nitrogen" OR "Nr" OR "reactive" OR "nitrous*" OR "NOX" OR "NO2" OR "ammonia" OR "NH3" OR "nitrate" OR "NO3" OR "fertilization" OR "long-term fertilization" OR "nutrient manipulation")
AND	

Valuation component	TS=("valuation*" OR "contingent valuation*" OR "choice experiment*" OR "non-market" OR "hedonic price*" OR "travel cost" OR "avoidance cost*" OR "replacement cost*" OR "WTP" OR "WTA" OR "socio-economic" OR "impact pathway*")
AND	
Value estimation	ALL=("cost*" OR "benefit*" OR "unitprice*" OR "externalit*")

Note: This search gave 698 results. Date of search: 12/12/2023

To assess the eligibility of papers during screening, the CIMO(S) search tool was employed to develop the following set of criteria:

- **(C) Context**: monetizing/valuing environmental impacts of (airborne) Nr deposition in sensitive terrestrial nature types in the Global North.
    - **Pollutants of interest:** airborne Nr, including NO_x, nitrous oxide (N_2O), NH_3, ammonium (NH_4^+) and nitrate (NO_3^-).
    - **Nature types of interest:** grasslands, forests, heathlands, peatlands, tundra, mires, fens, meadows, and sparsely vegetated areas.
    - **Region of interest**: Europe, North of the Alps
- **(I) Interventions of interest:** potential international agreements, EU policies, or national state laws regulating atmospheric emissions of Nr.
- **(M) Mechanisms of interest:** assess damage costs incurred by Nr deposition (e.g. €/kg NH_3), whereby the environmental impacts attributed to the emissions of a specific pollutant (e.g., kg NH_3/yr.) are quantified and valued through the entire sequence of the cause-and-effect pathway (*see IPA*).
    - **Impacts of interest:** damage to nature, including eutrophication from decreases in soil C/N ratio, overfertilization, and/or long-term fertilization, exemplified by losses in biodiversity (i.e. similar species richness).
- **(O) Outcomes**: derive WTP estimates (i.e. unit prices) for relevant environmental impacts, preferably from original preference-based valuation studies, which can be coupled to relevant emissions (e.g. €/kg NH_3).
- **(S) Study design:** preference-based valuation studies, while other valuation studies based on avoidance costs (i.e. shadow prices) or replacement costs methods (remedial/restoration) are also, although to a lesser degree, relevant.

The diagram in Figure C is a PRISMA flowchart that shows the steps of the review process. As depicted, a total of 2,276 studies (i.e. titles and abstracts) were imported into Covidence for screening, of which 438 were duplicates and subsequently removed. No references from other sources including grey literature (i.e. key-catalogues etc.) or citation searching were included in the review. Following the eligibility criteria presented above, a total of 1,790 papers were excluded during the title and abstract screening. The remaining 48 studies selected for an in-depth full-text review were schematized in an Excel sheet to assess their relevance for *Benefit Nature*. The spreadsheet included seven overall categories:

1. Study identities (e.g. author, year published/data collection)
2. Study characteristics and methods (e.g. IPA method, valuation study)
3. Geography (e.g. continent/country, target population)
4. Pollution assessed (e.g. pollutants, emissions media)
5. Impacts assessed/valued (e.g. nature types, eutrophication/loss of biodiversity/similar species richness)
6. Potential unit prices (e.g. cost/WTP estimates)
7. Payment characteristics (e.g. payment frequency, unit, vehicle)

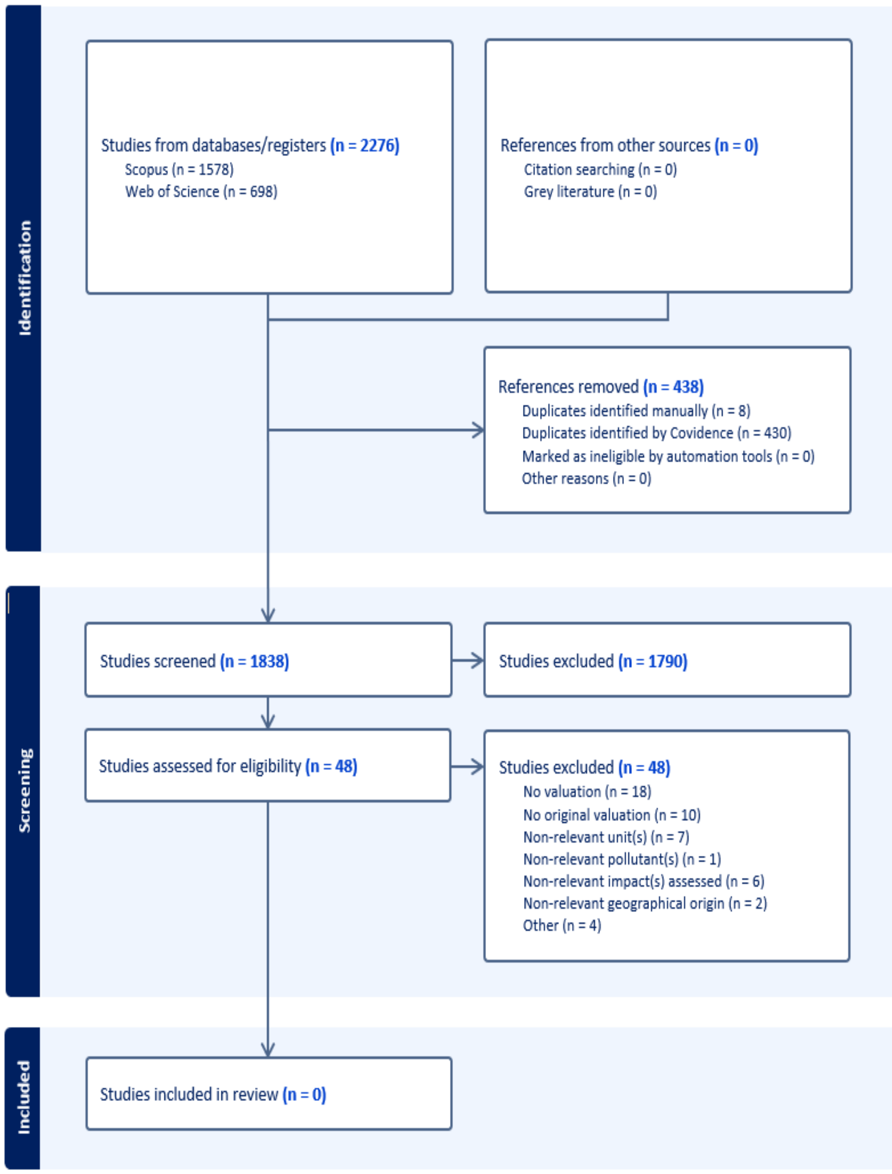

Figure C: PRISMA flow diagram for the valuation review. *Source: Imported from review in Covidence, 24/05/2024*

All 48 papers were for various reasons, as depicted in Figure C, excluded during the full-text review, the main reason being that they were not valuation studies per se or because they relied on value transfers, and thus didn't contribute with an original valuation. Note that we did not use so-called "snowballing" techniques to identify potentially relevant valuation studies (i.e. bibliography searching in papers, which rely on value transfers). The papers typically fall into the following categories:

- Only a few actual primary valuation studies, and studies that do address the value of environmental impacts of nitrogen pollution often rely on some sort of value transfer, making them ineligible.
- The few primary valuation studies often address impacts on human health, e.g. the EVA-health papers appear in the review.

- GLOBIO-model papers. The GLOBIO model (see e.g. Snipper et al., 2020) seems to have a similar scope to EVA-N and GLOBIO was originally *"... developed to quantify the impacts of infrastructure on biodiversity intactness (Nellemann et al., 2001), it was later extended to also include the impacts of climate change, land use (via both habitat loss and fragmentation) and atmospheric nitrogen deposition (Alkemade et al., 2009)."*
- Some IPA papers, although predominantly without quantified impacts in terms of valuation, which also include older and outdated papers, e.g. the EU project ExternE (Bickel & Friedrich, 2005).
- Quite a few papers are potentially relevant for the two other literature reviews on exposure-response and impacts pathways, concerning relevant pollutants (atmospheric nitrogen) and nature types (heaths and grasslands etc.).
- Quite a few papers with differing (not-directly relevant) scopes, i.e. valuation of non-relevant environmental impacts, non-relevant nature types, non-relevant pollution type/form/media, and non-relevant world region.
- Some socio-economic (cost-benefit) analysis but without original valuation, relying on value transfers.
- Some methodology papers on valuation methods and value transfer methods.

Summary of findings

The primary outcome of this literature review is the lack of truly relevant valuation studies in the scientific literature that apply appropriate methodologies to assess the environmental effects we aim to value. This may not be surprising, given the results in Magnussen et al. (2021) and Olsen et al. (2021) discussed earlier in this section. Nevertheless, this review represents an important step in *Benefit Nature,* because neither of the previous studies conducted a literature review with the same specific objectives or the same systematic and rigorous approach.

To some extent, our findings validate the conclusions of Magnussen et al. (2021) and Olsen et al. (2021). We did not identify any scientific studies that explicitly value the environmental effects of Nr deposition on Nordic ecosystems. This is noteworthy, considering that eutrophication of land and water has been recognized as an environmental problem for decades. The lack of recent studies may reflect the long-standing nature of the issue, which might have limited new research in this area.

The few studies that do align with the scope of our review are generally outdated, such as older working papers from the ExternE-project series, which are not suitable for our purposes. However, as Magnussen et al. (2021) highlighted, there may be potentially relevant studies within the "grey" literature. These studies often rely on benefit transfer methods and older primary valuation studies, which could still provide some insights in the absence of current scientific research.

Perspectives

After concluding the review, an additional search was conducted using a slightly modified search strategy to further verify the outcome of the review. Interestingly, two papers were identified as being genuinely relevant to the objectives of *Benefit Nature*, i.e. setting a methodology to value responses in sensitive natural land-based ecosystems from exposure to airborne Nr pollution.

The first paper, Christie and Rayment (2012), is a valuation study that examines the benefits of biodiversity within selected natural habitats in the UK. It employs a choice experiment (CE) to elicit WTP values for ecosystem benefits associated with Sites of Special Scientific Interest (SSSIs) (i.e., conservation areas). Among the benefits evaluated are biodiversity improvements related to achieving a ‘favourable’ condition, measured across seven ecosystem attributes, including (5) ‘Charismatic species’ and (6) ‘Non-charismatic species’. For a 1% increase in the provision of these attributes, the WTP/household/year was estimated to be (5) £2,49 and (6) £0,35, respectively. Note that although all attributes in the specified model were significant, (6) non-charismatic species were not. To estimate consumer surplus values per hectare for different habitat types (e.g., grasslands, heathlands, and bogs), a weighting matrix (WM) was developed based on expert input knowledge. This WM reflects the relative contribution of various SSSI habitats to the delivery of ecosystem services, linking WTP estimates for charismatic and non-charismatic species to specific habitat types. These habitat and nature types align with those relevant to *Benefit Nature*.

The second paper, Jones et al. (2018), develops the impact pathway for assessing responses in biodiversity from exposure to airborne Nr pollution. Using a value transfer approach, it applies the unit values derived by Christie and Rayment (2012) to calculate the marginal economic value of changes in Nr deposition. The study aims to create a spatially explicit methodology for quantifying impacts on biodiversity from Nr deposition, through specification of the steps in the IPA, schematized in the overview in Table C.

Table C: Overall methodology and findings in Jones et al. (2018)

IPA-step	Methods and results
Emissions/ deposition	Projecting changes in average Nr deposition in the UK between 2007 and 2020. • Using the FRAME (Fine Resolution Atmospheric Multi-pollutant Exchange) model, a Lagrangian atmospheric transport model used to assess the long-term annual mean deposition of reduced and oxidised nitrogen and sulphur (Smith et al., 2000). • Average Nr deposition was projected to decrease by 11%.
Dose-response relationship	Dose-response functions for Nr deposition and species richness in four nature types (heathland, acid grassland, dunes, and bogs). • Inverse relationship between deposition (kg N/ha/year) and plant species richness in 20 m^2 grids in all habitats. • Non-linear response function in all habitats except bogs, which indicates that the majority of biological impact on plant diversity occurs at relatively low levels of Nr deposition, but that it continues to have an impact at higher Nr deposition.
Impact of exposure	Changes in species richness due to Nr deposition measured as the diversity of non-charismatic species. • In response to a general decline of Nr deposition, there is a corresponding predicted increase in species richness. o At present it is not possible to model impacts of air pollution on charismatic species due to a lack of dose-response functions.
Valuation of impact	Calculation of the economic value of declining Nr deposition in terms of changes in the cultural ecosystem service ‘Appreciation of biodiversity’ (i.e. non-use value of non-charismatic species). • Value transfer function of unit values for non-charismatic species from Christie and Rayment (2012). • The marginal damage costs of pollution (i.e. the benefit to biodiversity per tonne decrease in Nr emissions):

	○ £103 per tonne of NO_2 not emitted (£33 to £237, 95% confidence interval) ○ £414 per tonne of NH_3 not emitted (£139 to £1.022, 95% confidence interval)

Jones et al. (2018) demonstrate that a value transfer approach can effectively link response functions for changes in species richness with WTP estimates for biodiversity, thereby establishing a clear impact pathway. As such, it highlights the significant economic benefits of policies aimed at reducing Nr emissions, but also that the spatial distribution of these benefits varies greatly between the selected habitats, recognized by the large confidence intervals. From a policy perspective, two key insights emerge from Jones et al. (2018). First, protecting habitats that are still largely unaffected by pollution, yields the highest economic values. Second, there are continued economic gains from further reducing Nr deposition in habitats already experiencing high levels of Nr pollution.

Achieving this outcome, however, involves several simplifying assumptions along the impact pathway. For instance, when estimating the impacts of Nr deposition, the study assumes that biological responses to changes in deposition occur within a year. This assumption overlooks potential delays in plant community responses due to factors such as the continued cycling of stored nitrogen in soil or species persistence effects, suggesting the need for a delay function to better capture these dynamics (Jones et al., 2018). Additionally, the analysis focuses exclusively on non-charismatic species, although perceived values for charismatic species are often far greater. This narrow focus could result in an underestimation of the costs associated with Nr deposition. Moreover, since Christie and Rayment (2012) noted that all ecosystem service attributes except non-charismatic species were statistically significant in their model, this introduces further uncertainty into the impact pathway for Nr depositions.

Building on the experiences from the literature review, the following section sketches an overall framework of how we see that the cost of biodiversity loss from air pollution can be assessed in the absence of (truly) relevant valuation studies.

Framework for inclusion of costs of biodiversity loss from air pollution

For complex goods like biodiversity, the economic theoretical basis for determining the value of a good is that humans obtain utility from their preferences for the outcome (i.e. change in that good). If that is the case, valuation estimates of biodiversity loss produced in the context of other pollutants can be used as an estimate for the loss. Further, it can be argued that if we are using the model to assess marginal changes, then a loss and a gain can be considered of the same value (Jacobsen and Lundhede, 2024). This means that valuation estimates of improvements in biodiversity conditions can be used to infer the value of biodiversity losses too. For the framework presented here, we use a similar approach as presented by Jacobsen and Lundhede (2024) and a per-species index (i.e. a WTP per species estimate). The overall approach is illustrated in Table D, which depicts the steps for calculating annual welfare loss and the loss in natural capital.

Jacobsen & Lundhede (2024) focus on red-listed species in a national (Danish) context. Here we are interested in all types of plant species as this corresponds to the exposure-response function. Hence, we use an estimate from Bakhtiari et al. (2018) assessing the general public's WTP for an increase in species diversity in a forest.

While some studies suggest differences in the value of different traits of biodiversity (Varela et al., 2018), there are also studies pointing at this being less important (Uggeldahl et al., 2025a; Jacobsen et al., 2012). Uggeldahl et al (2025b) finds that WTP for biodiversity in forests is slightly higher than for biodiversity in open landscapes, whereas Jacobsen et al. (2008) find small differences between habitats for species conservation. Thus, we do not correct any habitat differences.

Bakhtiari et al. (2018)'s estimate is 0.452 DKK/species/household/year for "a forest". While the size of the forest is not specified, the focus was on forests visited. Therefore, we use an estimate from Bue-Bjørner & Termansen (2014) who report that the average size of a visited nature area is 299.2 ha. Then the average WTP per species per hectare becomes 0.0015 DKK/household/year. Data was collected in 2012. Assuming an income elasticity of 1 and using the growth rate in nominal family income to inflation correct to 2023 prices, result in a WTP of 0.00202 DKK/species/ha/household/year. With 3,118,635 households (2023) this results in a WTP of 6,358 DKK/species/ha/year.

It can further be considered whether a delay function should be applied – reflecting a discrepancy (i.e. time lag) between when pollution changes happen and when the impacts on biodiversity are evident. For the current assessment, we have not made such a correction. Further, as opposed to the effect of air pollution on human health, which does not result in stock effects as pollutants are decomposed fast, there is an accumulation effect in that species that are lost, are also lost in the future. And similarly, when reappearing, they are also present the following years. Hence it can be argued that one should include the decline/increase in the stock of biodiversity in the estimate of the welfare economic loss. Using the same approach as Jacobsen and Lundhede (2024), for a declining discount rate of 3.5%, 2.5%, and 1.5% corresponding to the current recommendations of the Ministry of Finance in Denmark (Finansministeriet, 2023), result in a loss of natural capital of 704,172 DKK/species lost/ha. Table D below illustrates the calculation.

The loss in the natural capital stock is seen to be many times higher than the market price of land (a range of 100,000-300,000 DKK/ha is a reasonable estimate for that). On the one hand, this reflects the importance of biodiversity. But – it could be discussed whether respondents have sufficiently considered the spatial scale. The size of the forest is not estimated from the valuation study, but from another one. If we for example assume that people have considered a forest twice the size, the value estimate would be half the size. It is also likely that people have considered "a forest" to be "forests in the shown region". If that is the case, the loss in natural capital is around 700 DKK/ha/year. Another important assumption is the population included in the assessment. We used the entire Danish population. This assumes that all households in the country have utility of the species on a given site. Uggeldahl et al. (2025b) find that non-use values may indeed play a large role at a long distance and hence it is a reasonable assumption. But if that is the case, it could be considered whether the recalculation to a forest size of roughly 300 ha is appropriate. Hence, we believe the estimate would need further consolidation before being used in the EVA-N model system or similar models.

Studies with more direct weight on the spatial scale of the problem may be better for use in a context like this.

Table D: Illustration of the steps for calculating annual welfare loss and the loss in natural capital.

Element	Estimate	Unit	Source
WPT for 1000 extra species in a forest	452	DKK/household/year	Bakhtiari et al., 2018
WTP per one extra species in a forest	0.452	DKK/species/household/year	
Average size of visited nature area	299.2	ha	Bue-Bjørner & Termansen, 2014
WTP (2012) per species per ha	0.0015	DKK/species/ha/household/year	

Average annual growth rate in nominal family income 2012-2023	0.028		Statistics Denmark, 2025.
Annual WTP in 2023	0.0020	DKK/species/ha/household/year	
# households 2023	3,118,634		Statistics Denmark, 2025.
Aggregated annual WTP in 2023	6.4	DKK/species/ha/year	
Delay in species loss compared to pollution	0		Randomly set
Discount rate for delay function	0.035		Ministry of Finance (Denmark), 2025.
Annual welfare loss	**6.4**	**DKK/species/ha/year**	
Present value of future loss of species + growth rate correction	111	3.5%, 2.5%, 1.5%	Jacobsen & Lundhede (2024).
Loss in natural capital	**700**	**DKK/species lost/ha**	

Another avenue ahead that could be pursued is to use a nature intactness index and valuation hereof. Uggeldahl et al. (2025a, 2025b) argue that this may better capture the complex nature of biodiversity and people's value hereof. However, it would require an exposure-response function based on intactness rather than species, which may be even more challenging.

Review for Nr exposure-response functions

To search for relevant studies on Nr exposure-response functions of biodiversity, we used Web of Science, the search string given in Table E and the Covidence tool described earlier. We found 3,020 potentially relevant papers, where only 1 was a duplicate, bringing the total identified studies to 3,019 that were subsequently screened, based on title and abstract.

Table E: Exposure-response function review, WoS Search.

Search string	Search word
Nature type(s)	TITLE-ABS-KEY("natur*" OR "land-base*" OR "terrestrial*" OR "grassland*" OR "forest*" OR "heath*" OR "peat*" OR "sparsely vegetated*" OR "tundra*" OR "mire*" OR "fen*" OR "meadow*")
AND	
Pollution media/ effect/ component	TITLE-ABS-KEY ("air" OR "air pollution" OR "airborne pollution" OR ("atmospheric" AND "deposi*") OR "nitrogen" OR "Nr" OR "reactive" OR "nitrous*" OR "NOX" OR "NO2" OR "ammonia" OR "NH3" OR "nitrate" OR "NO3")
AND	
Plant exposure component	TITLE-ABS-KEY(("plant" AND "species richness") OR "similar species richness" AND "exposure" AND "response”)

Note: This search gave 3,020 results. Date of search: 2/2/2024

As biodiversity responses are likely to be quite location-specific at a global scale, the following criteria were used to identify relevant studies for the Nordic region:

i) Ecosystems from similar latitudes and climates as the Nordic countries (N-Europe/N-Asia/N-America and adjacent alpine areas).

ii) Contain data that allows extracting Nr exposure-response function of plant species richness as number of species per kg Nr additions per ha and year.
iii) Fertilization studies must have been conducted for at least 20 years.

The first screening of the 3,019 papers resulted in:

i) 2,896 papers (96%) were excluded because they did not contain usable Nr exposure-response data.
ii) 97 papers (34%) were excluded because of "wrong study design".
iii) 34 papers (1%) were discussion papers (no original data).
iv) 16 papers (0.5%) were excluded due to geographical area outside the area of interest
v) 11 papers (0.4%) were excluded because of "wrong settings"
vi) 2 papers (0.1%) were excluded because of "too short fertilization time period – wrong study design".

After completion of the initial screening, 26 papers were downloaded and read for further inspection. By inspecting their reference lists, 3 additional usable studies that had been missed in the first part of the review were added, bringing the total number to 29 studies.

18 papers of the 29 (62%) yielded usable Nr exposure-response functions for different ecosystem types. Table F shows median exposure-response functions derived from those found in the 18 papers. For final exposure-response function calculations, all the results need to be re-fitted with a comparable (linear) statistical model and as some datasets appear in more than one reference, those must be down-weighted in the final average response functions.

Table F: Median Nr exposure-response functions for plot-level species richness for different nature types and % change in the plot-level plant community (as derived from the preliminary analysis performed in the project).

Nature type	Species loss / 1 kgNr ha^{-1} yr^{-1} ADDITION*	References Nature type and country
Coastal dune ecosystems	-2.0 spp (-3%)	1) GB, 2) GB, 5) GB,
Heathlands	-0.3 spp (-1%)	1) GB, 2) GB, 3) GB, 7) GB, 8) GB, 11) GB, 16) GB, 17) GB
Acid grasslands	-0.5 spp (-1%)	1) GB, 2) GB, 3) GB, 4) DE, NL, GB, 12) GB, 13) GB, 14+15) DK, NO, SE, GB, BE, DE, IE, NL, FR; 16) GB, 17) GB
Calcareous grasslands	-0.0 spp (-0%)	3) GB, 16) GB, 17) GB
Bogs	-0.3 spp (-1%)	1) GB, 16) GB, 17) GB
Forests	-0.1 spp (-1%)	6) FI, SE, PO, AT, DE, 10) USA, 16) UK, 17) UK
Alpine areas	+0.1 spp (+0%)	9) CH; 18) FR, ES

* Loss at an increasing rate of Nr-deposition is more-or-less instantaneous; spp additions when Nr deposition decreases come with a delay function that is at present highly uncertain; 1) Jones et al. (2018), 2) Field et al. (2014), 3) Maskell et al. (2010), 4) Dupré et al. (2010), 5) Jones et al. (2004), 6) Dirnböck et al. (2014), 7) Payne et al. (2014), 8) Caporn et al. (2014), 9) Roth et al. (2013), 10) Simkin et al. (2016), 11) Southon et al. (2013), 12) Stevens et al. (2004), 13) Stevens et al. (2006), 14) Stevens et al. (2010), 15) Stevens et al. (2011), 16) Tipping et al. (2013), 17) van den Berg et al. (2016), 18) Boutin et al. (2017)

In the process of reviewing, we realized that there are comparably many studies presenting results for the UK. This means that the resulting exposure-response functions cannot avoid being biased towards UK nature and environmental variables. Luckily, the UK nature profile is similar to what is found in Northern Europe and Southern Scandinavia, and we therefore have not taken steps to mitigate this issue. We have prioritized not to include studies from e.g. China and South America, because although nature types with the same names exist, the species composition in the corresponding nature types in these areas, differs greatly from the composition in Scandinavia and Northern Europe. Further, we have only used forest and woodland data from USA, as those ecosystems have similar understories to their European Boreal and Temperate forest counterparts.

Based on the results in Table F, it is possible to estimate loss in plant biodiversity (species richness) in different habitats when Nr deposition increases. The included studies in Table F all represent habitats (nature types) similar to those found in Denmark (see Table I) and their predicting power should therefore be relatively strong for Danish conditions. It is, however, noteworthy that only one of the references used to establish such Nr exposure-response functions included data from Denmark, even if both deposition (Ellermann et al., 2024) and plant biodiversity monitoring (Ejrnæs et al., 2021) studies exist.

We also discovered other obstacles during the work carried out in the *Benefit Nature* project. Almost no field studies seem to exist, where reductions in Nr deposition have been related directly to changes in plant biodiversity. This is very unfortunate, as the response to decreasing Nr inputs is unlikely to be the same as the response to increasing inputs (Storkey et al., 2015). This is due to the fact that the negative effect of increasing Nr inputs on plant biodiversity to a large extent is driven by "a fertility response" that is likely to remain as long as the Nr added in the past remains active in the plant-soil nutrient cycle. There are some modelling studies from the UK that have addressed this by introducing 30-year "running average" in Nr-inputs, when deposition is decreasing (Payne et al. 2017; 2019), but overall there is a great need for empirical studies monitoring what actually happens in nature, when the Nr deposition decreases. As Nr deposition has been decreasing in Denmark in the past couple of decades (Ellermann et al., 2024), this is of high importance for the current and future development of the EVA-N model system.

Review for model systems using the impact-pathway approach

The purpose of the review for other studies using the IPA is to qualify and discuss the decisions made during the design of the EVA-N system. Using Web of Science and the search string given in Table G, 3,413 studies were identified. Of these 1,559 were duplicates identified by Covidence (the above-described tool applied for conducting the reviews), leaving 1,854 studies for the first screening, based on title and abstract.

Table G. Impact-pathway approach review, WoS Search.

Search string	Search word
Nature type(s)	TITLE-ABS-KEY("nature*" OR "land-base*" OR "terrestrial*" OR "grassland*" OR "forest*" OR "heath*" OR "peat*" OR "sparsely vegetated*" OR "tundra*" OR "mire*" OR "fen*" OR "meadow*")
AND	
Pollution media/ effect/ component	TITLE-ABS-KEY ("air" OR "air pollution" OR "airborne pollution" OR ("atmospheric" AND "deposition" AND "nitrogen") OR "Nr" OR "nitrous*" OR "NOX" OR "NO2" OR "ammonia" OR "NH3" OR "nitrate" OR "NO3")
AND	
Plant exposure component	TITLE-ABS-KEY(("plant" AND "species richness") OR "similar species richness" OR "species richness" AND "critical load exceedance" OR "critical load" OR "impact-pathway" OR "impact pathway approach" OR "impact assessment" AND "valuation" OR "economic value" OR "contingent value" OR "avoidance cost*" OR "damage cost*" OR "socio-economic")

Note: This search gave 3,413 results. Date of search: 14/03/2024.

Of the 1,854 remaining studies, 1,793 were considered irrelevant based on screening of title and abstract, giving a number of 61 studies to be examined in full text. Of these 61 studies, another 56 were excluded based on *wrong study design* (30), *no economic valuation* (11), *not focused on* *sensitive nature* (6), *wrong pathway* (5), *methodology paper* (3), *wrong outcomes* (1), leaving 5 papers included for extraction:

1. Jones et al., 2018
2. Wagner et al., 2017
3. Feng et al., 2021
4. Song et al., 2023
5. Rowe et al., 2017

Of these five papers (presented in more detail in Table H), one had a broader focus on creating an accounting framework for pollution damage cost estimation, covering pollutants in general rather than nitrogen specifically. One study focused on reviewing existing metrics for nitrogen pollution and creating a new integrated one. The last three papers extracted, analysed the steps in the impact-pathway chain in slightly different ways, with either Nr deposition or NH_3 emissions as the primary focus area. All in all, the 5 extracted papers will contribute to the understanding and qualification of the elements of the EVA-N system, when the system is further developed at a later stage. However, for the moment, given the issues with no explicit valuation functions and uncertainty in delay functions for exposure, it is too early to explore in depth the consequences for our system of the findings of the five studies.

Table H: Overview of methodologies, research focus, and potential biases in the reviewed studies using IPA.

	Methodology	**Focus**	**Risk of bias**
1	Spatially explicit approach to assess nitrogen's effect on biodiversity, specifically species richness in habitats vulnerable to nitrogen deposition (e.g., grasslands, heathlands, dunes, bogs). The study develops dose-response functions to measure how nitrogen deposition alters species richness. Employs **value-transfer methods** based on WTP for "appreciation of biodiversity" as the primary ecosystem service. Valuation is focused on biodiversity improvements associated with reduced nitrogen levels, providing a monetary assessment of public appreciation for biodiversity.	The sole focus on the non-use value of biodiversity limits the valuation pathway by excluding other ecosystem services that may also be affected by nitrogen deposition.	The reliance on "appreciation of biodiversity" (in terms of WTP for non-charismatic species) as the sole valued service may create bias by underestimating nitrogen's broader ecosystem impacts.
2	Utilize the **Potentially Disappeared Fraction (PDF)** approach to quantify biodiversity impacts from NH_3 emissions in Lower Saxony, Germany. The PDF model estimates the proportion of (target) species potentially lost due to nitrogen pollution, creating a direct link between nitrogen levels and biodiversity metrics. On this basis, a **cost-benefit analysis (CBA)** is conducted by comparing the costs of NH_3 abatement for farmers against the societal benefits of reduced biodiversity and health damage costs. This model integrates biodiversity loss with broader social costs, offering a holistic view of nitrogen's impact on ecosystems and human well-being. Total societal benefits exceed abatement costs, suggesting implementation is justified.	Focuses on landscape-level impacts, however without single-species focus or ecosystem specificity, it lacks detailed sensitivity for highly impacted habitats.	The PDF approach is effective for broad biodiversity valuation but may generalize ecosystem responses, potentially missing detailed responses in sensitive habitats.
3	This review presents an **accounting framework** for pollution damage cost estimation, covering pollutants generally rather than nitrogen specifically. The authors argue that pollutants impact the environment at various spatial and temporal scales and that therefore it is important to account for this spatial-temporal heterogeneity.	Lacks a direct focus on nitrogen's impact on biodiversity or specific sensitive ecosystems, thus providing only a broad framework. The latter may have limitations in accounting for habitat- or species-specific response, also	Lack of specificity in sensitive ecosystems limits the relevance for nuanced biodiversity assessments.

		due to the scarcity of studies covering the effects of pollutants on species.	
4	Employ a **flow analysis model** to quantify Nr emissions in the Yellow River Basin and applies a **CBA** to assess the economic costs associated with nitrogen pollution, including biodiversity damage and aesthetics. The economic valuation focuses on damage costs in form of human health costs (health impacts from NH_3, NOx, and N2O emissions, as well as groundwater pollution), environmental damages (costs related to biodiversity loss, increased eutrophication in surface waters, and aesthetic losses due to reduced visibility) and climate change (anticipated damages from greenhouse gas emissions).	The study provides an aggregated analysis of nitrogen's effects across subsystems like forests and grasslands without a targeted pathway focused on single-species or sensitive ecosystems. The impact pathway is broad, quantifying environmental aesthetics alongside general biodiversity loss.	Lack of specificity in sensitive ecosystems limits the relevance for nuanced biodiversity assessments.
5	The study evaluates existing metrics for nitrogen pollution (pressure, midpoint, and endpoint metrics for N pollution) and proposes a new integrated one. It discusses the ecological implications of nitrogen pollution in sensitive ecosystems such as grasslands, utilizing metrics like species richness and nitrogen leaching to evaluate biodiversity impacts. While valuable for understanding ecological factors, the study lacks economic metrics.	Establishes a strong pathway between nitrogen levels and biodiversity loss using ecological indicators, but without an economic dimension, it does not assign monetary values to ecosystem impacts.	Strong in ecological indicators but lack of economic translation limits its relevance for cost-benefit analysis.

Synthesis and ways forward

All studies indicate nitrogen's detrimental effects on ecosystems, highlighting the need for improved nitrogen management to preserve biodiversity. Jones et al. (2018) and Wagner et al. (2017) provide pathways for linking nitrogen impacts to economic valuation. In contrast, Feng et al. (2021) and Song et al. (2023) offer broader, less ecosystem-specific methods, while Rowe et al. (2017) focuses exclusively on ecological impacts. A limited ecosystem service valuation may not fully capture biodiversity's broader economic contributions. Integrating a more comprehensive set of ecosystem services could enhance the valuation pathway (compare Jones et al. (2018) with Song et al. (2023)).

Ways forward in the development of EVA-N (and other IPA-based model systems) could be:

- **Expanded ecosystem service valuation**: Future research could incorporate multiple ecosystem services associated with biodiversity (loss). This would capture biodiversity's multifaceted value beyond "appreciation," enhancing the impact pathway assessment for sensitive ecosystems.
- **Balancing species-specific and general biodiversity frameworks**: On the one hand, adapting models with respect to single species in sensitive habitats could improve sensitivity, offering a clearer link between nitrogen reduction and specific biodiversity outcomes. Thus, offering detailed information for conservation targets, as they can indicate specific ecosystem functions and services. On the other hand, focusing on broader aspects like species richness can provide a more comprehensive understanding of ecosystem health, as it captures overall biodiversity changes. Thus, it can better reflect biodiversity's vulnerability to nitrogen impacts. A combined approach that considers both species richness and individual species impacts may offer the most effective biodiversity assessment.

EVA-Nature system development

The second part of the Benefit Nature project focused on developing a new system – EVA-N – that can be used for impact assessment of costs and benefits for sensitive terrestrial nature in relation to Nr deposition. The steps included in the EVA-N system were presented in Figure A, starting with emissions in the top left corner of the figure, which serve as input to the air pollution model DEHM, which calculates the distribution of concentrations and depositions of a number of air pollutants, including all the components relevant for the spatial and temporal distribution of Nr deposition. Combining DEHM results with detailed land cover data, the Nr deposition to nature areas is obtained (step 3 in the center of the figure), and this can then be combined with exposure-response functions for the specific nature area type, to estimate the effects on biodiversity (here identified as similar plant species richness, as described earlier). The final step includes the economic valuation of species decline/increase to obtain an estimate for the socio-economic costs. Since it has not been possible to recommend values for plant species richness decrease/increase, this last step is not yet included in the EVA-N model system

The emissions of NH_3 contribute significantly to Nr deposition, both through dry deposition of NH_3 itself, which happens relatively close to the sources, and through deposition of ammonium (a reaction product in the atmosphere from NH_3), which happens also further away from the sources. Denmark is characterized by a large activity level in the agricultural emission sector, with the highest level of emissions typically taking place on the most nutrient-poor and sandy soils, i.e. in the Western part of the country. Also, the proximity to large source areas in Northern Germany and the Netherlands influences the concentration and deposition of Nr in Denmark. In Figure D, this pattern can be observed for NH_3 in the plots, which show the annual total emission of NH_3 obtained from the national emission inventory for 2019 (and included in the model calculations) in the left panel, and the resulting annual mean concentration and total dry deposition of NH_3 for the Danish area in the center and right panel, respectively.

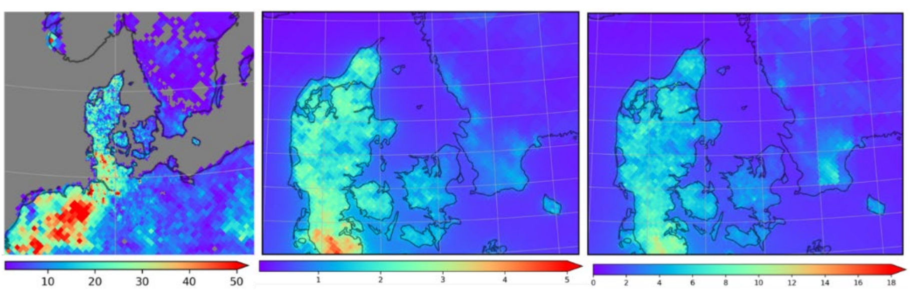

Figure D: Annual emission of NH_3 [kgN/ha] (left), annual mean concentration of NH_3 [μg/m^3] (center), and annual total dry deposition of NH_3 [kgN/ha] (right) for 2019. The emission data serve as input to the DEHM calculations, whereas the concentrations and depositions are results of the calculations.

Comparing the pattern of NH_3 concentration and dry deposition of NH_3 (Figure D center and right) with the extent of Danish terrestrial dry and wet nature areas in Figure E, it can be seen that there is some overlap between sensitive nature areas and areas that have a higher than average exposure to Nr concentration levels and deposition load.

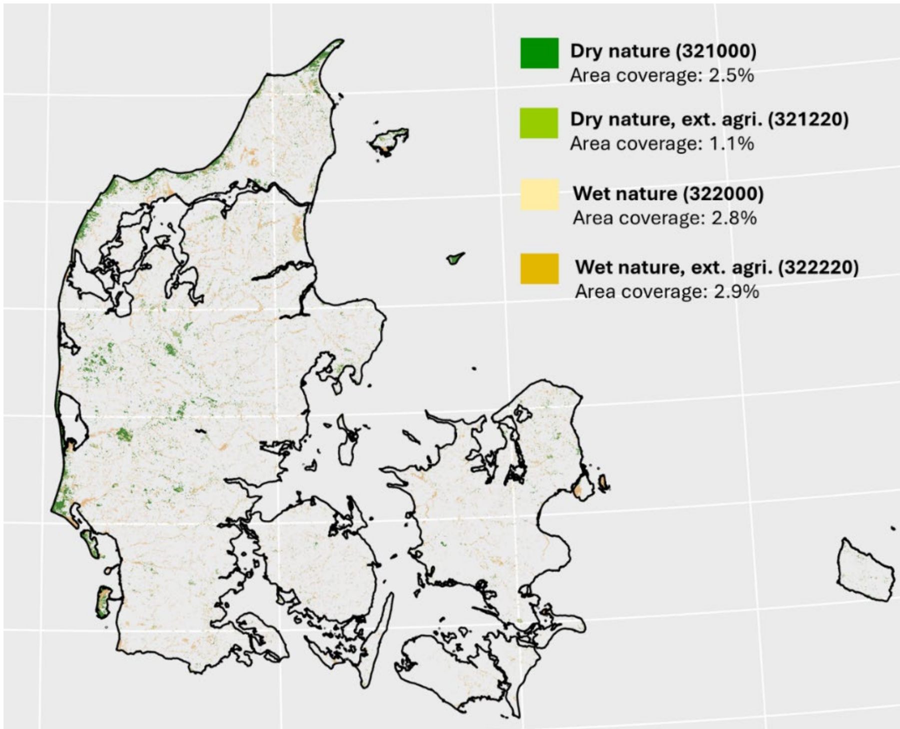

Figure E: Land cover data from Basemap04 for 2021 for four aggregated land cover classes that constitute all the terrestrial nature areas in Denmark. Source: Levin et al., 2022, https://dce2.au.dk/pub/TR252.pdf.

The spatial resolution of the annual Nr deposition results from the DEHM model is 5.56 km x 5.56 km, however, the actual magnitude of the Nr deposition to a specific area depends on the properties of the surface and is related to vegetation characteristics (e.g. leaf area index, stomatal conductance) and meteorological conditions (e.g. temperature, relative humidity, turbulence, and wind speed). DEHM calculates the magnitude of the Nr deposition to all the different types of surfaces within each model grid cell, and coupling the data from DEHM with the land cover information from Basemap04 in the EVA-N system enabled us to estimate the land-cover-weighted annual Nr deposition to the total areas of specific nature area types. An overview of the selected nature area types and the area they cover in Denmark is given in Table I.

Table I: Nature types (dry and wet) and area covered by these (in hectares) according to the definition from the Basemap04 data.

Nature type	Area (hectares)
Heath	73,045
Grassland	43,954
Acid grassland	3,795
Calcareous grassland	1,269
Dry nature areas, beach	6,834
Dunes	22,485
Other vegetated areas (juniper scrub, forest edge, etc.)	2,092
Nonvegetated (cliffs, fire belts, etc.)	1,543

Wet nature areas, beach	44,343
Meadow	105,472
Bogs (incl. raised), mires and fens	92,570
Total	397,402
Total dry	155,017
Total wet	242,385

To examine the consequences of Nr deposition to Danish nature areas, time series of Nr deposition to Danish land surfaces were established with the DEHM model for the period 2010 to 2021. Based on the Nr deposition data for this period, changes in similar plant species richness for dry and wet nature types were calculated with the EVA-N system, using the exposure-response functions derived within this project. Figure F displays the annual total Nr deposition weighted across all nature areas in Denmark of the *heath*-, *grassland*- and *meadow*-type, respectively, for the years 2010-2021. It can be seen that the level in 2021 is almost the same as the level in 2010, with some changes from year to year in between.

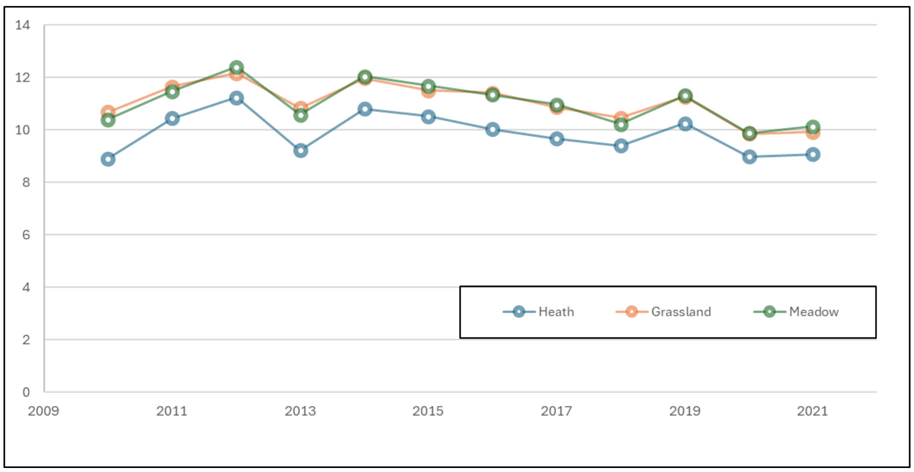

Figure F: Annual total Nr deposition [kgN/ha] calculated with the DEHM model and weighted across three types of nature areas in Denmark using the Basemap04 data for the period 2010 to 2021.

From the review of exposure-response functions described above, we know that there is a relationship between an increase in atmospheric Nr deposition and a decrease in similar plant species richness, but the changes in plant species richness following the decrease in atmospheric Nr deposition, are much more uncertain. A recent study (Payne et al., 2019) analyzed what the most meaningful metric of atmospheric Nr deposition for understanding observed changes in plant species richness was, and the results suggested time scales of around 30 years.

To analyze and better understand what the response from changes in Nr deposition could look like, we assumed delay functions of 5 and 30 years, respectively, for increase and decrease in Nr deposition and examined what happens for hypothetical examples of dry and wet nature areas giving the time series established for Nr deposition for 2010-2021. The results are presented in Figure G, showing the evolution over time until 2050 in plant species richness for a generic dry (Figure G, left) and wet (Figure G, right) nature area following the inter-annual increases and decreases in Nr deposition calculated with the DEHM model for Denmark for 2010-2021. The plant species richness has been normalized to 100 in 2010, and the response in plant species richness after 2021 reflects the increases in Nr deposition for the first five years, and the decreases in Nr deposition for the next 25 years, due to the difference in delay time of the response. All nature areas recover from the exposure in this simulation, but the response to the changes in Nr deposition within the time period from 2010 to 2021, is seen to have consequences for many years after. So even though the deposition is quite similar over the 12 years from 2010 to 2021 (Figure F), the response in the ecosystem amounts to an intermediate species loss of up to 12%.

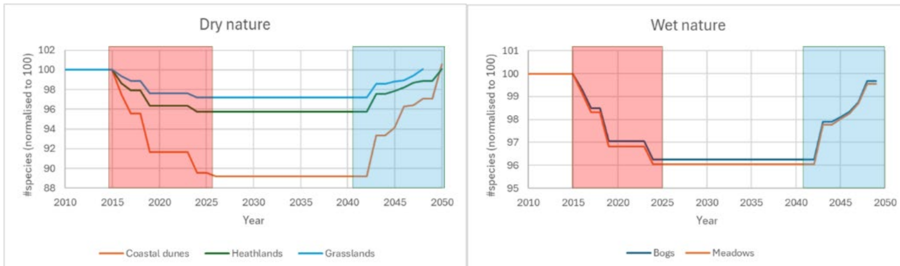

Figure G: Normalised response in generic dry and wet nature areas to inter-annual Nr deposition changes from 2010 to 2021. Red shaded areas correspond to the time of response from increases in Nr deposition, blue shaded areas correspond to the time of response following decreases in Nr deposition.

Conclusions and future work

The first question raised in the *Benefit Nature* project was: Is it possible to assess the socio-economic costs and benefits of **terrestrial biodiversity impacts** in a way similar to health impact assessments?

The overall result of the literature review on valuation is that we did not find any truly relevant valuation studies in the scientific literature that use the proper valuation methodology and values for the environmental effects of Nr deposition. This was not surprising, given the results of the reviews by Magnussen et al. (2021) and Olsen et al. (2021). However, this review was still an important step in highlighting the need for new valuation studies in this research area. Moreover, as highlighted by Magnussen et al. (2021), there may exist some potentially relevant “grey” literature, although these often rely on value transfers of older primary valuation studies.

We also find that other researchers (e.g. Christie and Rayment, 2012; Jones et al., 2018) have tried to develop the impact pathway for nitrogen pollution in the UK. Their results suggest that it is possible, although a complex task, to assess the socio-economic benefits of terrestrial biodiversity due to controls in atmospheric nitrogen pollution, in a way similar to impacts on human health. Building on the experiences of the review and adopting a similar approach as presented by Jacobsen and Lundhede (2024) (i.e. green net national income approach), we presented a preliminary framework for valuing biodiversity loss from air pollution in Denmark. We also highlighted the challenges in developing this framework and the “strong assumptions”, which, for now, makes it advisable not to use the calculations in the EVA N-system.

Further, we find that it would be theoretically consistent to use estimates of biodiversity loss and gains caused by other processes than air pollution. However, little is known empirically of whether the means of change matter. For future research, we recommend that it is further investigated whether an assumed independence between the means and the resulting values in valuation holds for air pollution.

The second question raised was: Are there **exposure-response functions** to be used?

In the second literature review focused on exposure-response functions, we managed to find 18 scientific papers that could contribute to a first prelimineary set of exposure-response functions for the ecosystem types we selected for analysis. The functions were in different units and for different setups, but we have converted them to reflect the loss of similar plant species from a one kg addition of Nr pr hectare per year, and the similarity between the functions after the conversion was high. The set of functions developed here are biased towards Great Britain (GB) in the way that all of the available studies of exposure-response for the ecosystem types *coastal dunes*, *heath*, *calcareous grassland* and *bogs* were of GB origin. For the ecosystem type *acid grasslands*, the majority of studies were from GB, but there were also other sources, and for the ecosystem types *forest* and *alpine areas*, there were only studies available from outside GB. All studies were from European countries, except one study for *forest* which was obtained from USA ecosystems.

So the conclusion is that there *are* exposure-response functions available that can be used for assessment of impacts of atmospheric Nr deposition. One major constrict is that all studies refer to situations, where the Nr deposition is increasing, and as discussed earlier, this means to display great caution if attempting to use these functions for Nr deposition decreases, because of the differences in lag-time for the response.

The third question was: How can changes in sensitive nature status/biodiversity be **valued**? And what is the **metric** for response and valuation?

It is important that the valuation estimates are based on the same metric as the exposure-response function. In our approach, the latter is based on species presence – in fact plant species. Using the same approach as used in a project developing a net national income measure for Denmark, we used estimates assessing the marginal value of a species increase/decrease to assess the value of the response from the exposure-response function. We assess that this is a consistent approach, but the estimate used may not sufficiently capture the spatial scale of the problem. Hence, if we are interested in per-hectare measures as here, we need to have valuation studies also being dependent on that. Other approaches for valuation could be used and might better capture the value (e.g. biodiversity intactness), but then the exposure-response function could not be used. Hence it does not seem like a viable approach at this stage.

In the valuation estimate, we came up with two values – an annual loss expressing the annual loss of a species and a loss in the natural capital value. The annual loss captures the loss of species that we do not have today. The loss in natural capital value captures the loss to future generations of the loss occurring today. Hence, if a species is disappearing it causes a loss today – and in future generations. Likewise, if species are reappearing, it will include the gain to future generations of such restoration.

The scale of the problem in the valuation study we based the calculation on, may not be sufficiently precise to produce with an estimate. But with the use of a per species valuation estimate with explicit consideration of the spatial scale, we find that it is a possible avenue ahead.

The last question raised was: “Would it be possible to help answering questions like: What **measures** are the most relevant in the ***agricultural green transition*** to provide the largest **benefits** from reduction of reactive nitrogen deposition?”

In the current study we have not looked at spatially specific estimates. But once included in the air pollution model, it would indeed be possible to answer such questions.

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Appendix A The impact-pathway approach (IPA)

The impact-pathway approach (IPA) provides a systematic framework for assessing the impacts of pollution on the environment and human health. When applying an IPA to a pollution problem, the impacts that can be attributed to the emissions of a pollutant are evaluated through the entire sequence of the cause-and-effect pathway. The fundamental steps of the IPA are illustrated in Figure A.1, beginning with; 1) emissions at the source; 2) models of dispersion through the environmental medium to estimate concentration at receptor sites (i.e. humans or the environment); 3) dose-response functions to assess impacts or toxicity of exposure; and 4) economic valuation of impacts at the endpoint. The methodology was developed as a part of the European research project, ExternE (Externalities of Energy), which had the objective to assess human health impacts of air pollution originating from energy and transportation (Bickel & Friedrich, 2005, p. 7).

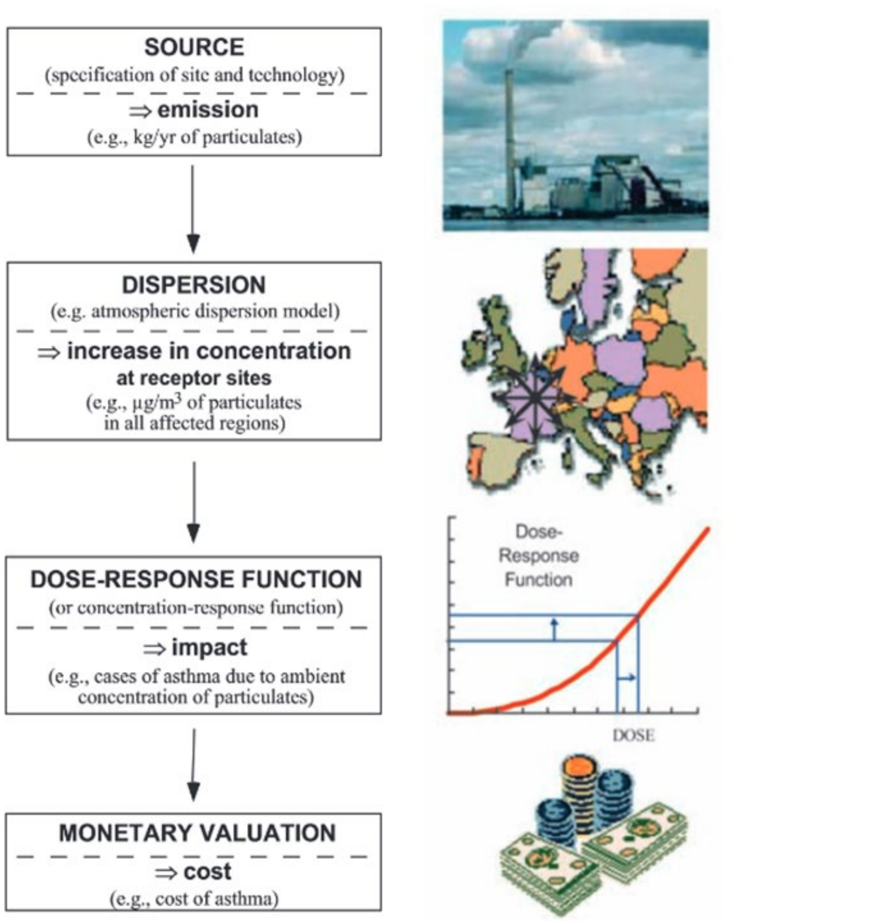

Figure A.1: The principal steps in the Impact Pathway Approach (IPA) with airborne pollution as an example.

Source: Bickel & Friedrich (2005), p. 2.