



Exmoor Air Quality Management Strategy: Evidence and measures

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0. Glossary

SSSI – Site of Special Scientific Interest

SAC – Special Area of Conservation

AQMS – Air Quality Management Strategy

CL – Critical Load defined as: “a quantitative estimate of deposition of one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge” (Nilsson and Grennfelt, 1988).

CL_e – Critical Level, for vegetation, are defined as: “concentration, cumulative exposure or cumulative stomatal flux of atmospheric pollutants above which direct adverse effects on sensitive vegetation may occur according to present knowledge” (ICP Vegetation, 2017).

Hysteresis - where the state of a system depends not only on its current conditions but also on its past history. where the same environmental conditions have different ecosystem properties associated with them due to internal system feedbacks that prevent return to the previous (desired) state (see e.g. Maes et al. (2024)).

1. Introduction

Exmoor National Park is home to temperate rainforests, internationally important lichen communities and a rich cultural heritage of farming and land management. This unique ecosystem thrives partly due to a long history of traditional land management practices that have preserved its biodiversity and supported a rich diversity of habitats. Characterised by hilly open moorland with areas of woodland, the national park spans over 692 km² and features 15 Sites of Special Scientific interest (SSSIs) and 2 Special Areas of Conservation (SACs). However, atmospheric nitrogen-based air pollution threatens this landscape by putting pressure on the sensitive designated features which are protected within SSSIs and SACs in the National Park.

Atmospheric nitrogen (N) deposition and elevated ammonia (NH₃) concentrations are significant threats to sensitive habitats and species. Exposure of sensitive vegetation to atmospheric N can result in acidification and eutrophication and lead to declining populations of key species of high conservation value. This is because they are replaced with faster growing species that are able to exploit the conditions of higher nitrogen input and outcompete the N sensitive species (e.g. Dise et al. 2011). Both SACs within the National Park, Exmoor Heaths SAC and Exmoor and Quantock Oakwoods SAC, are estimated to be in exceedance for the critical thresholds for nitrogen-based air pollution, i.e. critical levels associated with



ammonia, and critical loads associated with nitrogen deposition. Therefore, air quality related to atmospheric N is considered a threat to both sites.

To manage the effects of atmospheric N pollution in the National Park, it is crucial to focus on sustainable management at the local level to complement action taken regionally, nationally and internationally. Therefore, Exmoor National Park Authority have commissioned the development of an Air Quality Management Strategy (AQMS). The aim of the AQMS is to reduce atmospheric N pollution below critical thresholds in protected areas of the national park (SACs and SSSIs) and mitigate existing sources as well as avoid future emission sources in these areas. The strategy will outline a pathway for the collective management of local scale atmospheric nitrogen pollution for the protected areas in Exmoor, identifying practical and deliverable measures to reduce current pollution levels. It is expected to be part of a wider co-ordinated effort, with local, regional and national stakeholders coming together to reduce the N pollution threat in Exmoor National Park.

As part of the AQMS development, the project team has collated evidence to characterise and understand the patterns and levels of atmospheric N pollution across Exmoor and identify key emission sources impacting the protected areas. The evidence base includes a combination of data and maps on air quality, a review of the latest scientific literature (where relevant) and information on land management from local stakeholders. This report summarises the available evidence and assesses the atmospheric N threats to both the National Park as a whole and the 2 SACs and 15 SSSIs within Exmoor National Park.

2. Datasets & Methods

To understand the key atmospheric N threats and dynamics at each designated site, recent UK national scale modelling results and datasets were analysed. The datasets included high-resolution gridded model estimates of agricultural and non-agricultural NH₃ and NO_x emissions, NH₃ concentrations, N deposition and dominant NH₃ emission sources. All modelled datasets are in line with the UK's national datasets (National Atmospheric Emission Inventory¹; assessment of critical loads and levels exceedance as published in the annual Defra Trends reports, e.g., Rowe et al. 2024). NO_x concentrations were not considered as the UK Air Information Resource modelling (UK AIR²) indicated they were significantly below the current critical level (CLE) in all areas. NO_x emissions were considered in order to understand their contribution to N deposition.

The following key datasets were used in this assessment:

¹ <https://naei.beis.gov.uk/>

² <https://uk-air.defra.gov.uk/data/modelling-data>



2.1 Modelled atmospheric N data

Agricultural and non-agricultural emissions: The latest published National Atmospheric Emissions Inventory (NAEI) maps available were for the year 2021. These are annual maps of agricultural and non-agricultural emissions by source sector, at a 1 km x 1 km grid resolution level maps. It is important to note that these modelled emission estimates are based on national-scale data and assumptions. For agriculture, the data are based on average management practice for different sectors and subsectors, as there are no data on individual farm practices available for this modelling. The data are smoothed out at a grid resolution of 1 km x 1 km, meaning that local emission hotspots (such as slurry stores or livestock houses) may be present but not shown in the gridded data. Additionally, to preserve confidentiality of some underlying licensed datasets, data are further aggregated so that no individual farm holding's data can be back-calculated from the maps.

NH₃ (ammonia) concentrations: The latest three-year average modelled NH₃ concentrations (2019 – 2021) were used in this study. These were modelled using the EMEP4UK atmospheric chemistry and transport model. EMEP4UK is a UK version of the EMEP MSC-W atmospheric chemistry transport model (<https://www.emep.int>). NH₃ concentrations calculated with EMEP4UK are calibrated against the National Ammonia Monitoring Network (NAMN), developed and operated by UKCEH, for calculation of exceedance of the NH₃ critical levels. The median bias in the model-measurement comparison was used to bring modelled concentrations in line with measured values. This approach is the same as that used in the annual Defra Trends reports (e.g. Rowe et al. 2024).

N (nitrogen) deposition: A gridded three-year mean (2019-2021) Concentration Based Estimated Deposition (CBED) model estimates of N deposition were summarised to show the range of N deposition (min, max and area-weighted mean) for Exmoor NP as a whole and each designated site. The data provide a breakdown of wet and dry deposition, as well as oxidised and reduced nitrogen, i.e. deposition originating from NO_x emissions (mainly combustion) and NH₃ emissions (mainly agriculture and waste management).

The CBED model uses various inputs (including an ammonia concentration map generated using the EMEP4UK model, as discussed above) and produces deposition estimates at a 5 x 5 km grid. These were downscaled to a 1 km x 1 km grid using bi-linear interpolation. To even out year-to-year variation in meteorology, all deposition data presented are three-year averages (2019-2021).

UK source attribution dataset: The main emission sources contributing to N deposition at each site were estimated using modelled source attribution data for the year 2018 (as also available via apis.ac.uk). The data are derived by performing multiple model runs of the EMEP4UK model, with each source type/category (such as agriculture as reported by the NAEI) or individual large point sources (such as power stations) removed in turn. Each of these model runs is compared against the original run (with all sources) to determine the proportion the missing source



category contributed to total N deposition. This allows for the N deposition footprint attributed to these individual emission source categories to be calculated as a proportion of total deposition to each model grid square. The dataset was summarised to quantify key emission sources of atmospheric N that contribute to deposition at each designated site and for the National Park as a whole.

Dominant agricultural emission sources are calculated as NH₃ emissions sources which contribute > 45% of total NH₃ emissions. This 45% threshold was selected to optimise dataset interpretability – a 50% cut-off would result an overrepresentation of mixed-source categories, while 45% maintained majority dominance in most grid square while improving interpretation. Using data from 2020, determination of dominant sources is done sequentially, starting with the highest resolution (most specific) categories (e.g., beef cattle). If these higher resolution categories do not exceed the 45 % threshold for dominance, categories are combined to create increasingly general categories (e.g., beef cattle < mixed cattle < mixed agriculture). This aggregation was implemented to enhance interpretability, as broader categories provide clearer patterns when specific sources fall below the dominance threshold. To ensure individual holding level data are not disclosed, each category must contain at least five individual farm holdings. This is also true for the agricultural emission density estimates, where categories of < 5 holdings are amalgamated into the ‘other agricultural sources’ category

N Futures modelling. JNCC Nitrogen Futures Project developed and modelled scenarios of different policy targets and options until 2030/2040. The following scenarios will be summarised for the two SACs and Case Study areas in this report.

- **Baseline** – A modelled estimate of present-day conditions (~2017 at time of modelling)
- **2030 Business As Usual (BAU)** – An estimate based on the implementation of what currently agreed/in progress national policies will achieve. This scenario includes no spatial targeting.
- **2030 NAPCP** – This scenario is based on achieving the UK-wide emission reduction target set by the National Emission Ceiling Regulations (NECR), as outlined in Defra’s National Air Pollution Control Programme (NAPCP). The NAPCP was extrapolated from England to the rest of the UK and emission reductions were disaggregated across the four UK countries, with each expected to contribute a roughly proportional share. To reflect differences in agricultural systems and policy contexts, each Devolved Administration was then consulted to develop a realistic bundle of measures for each country based on the systems in place and likely implementation rates within their jurisdiction.

2.2 Monitoring data

The Braunton Burrows National Ammonia Monitoring Network site (NAMN; https://uk-air.defra.gov.uk/data/non-auto-data?network=namn&uka_id=UKA00969)



NH₃ concentrations were analysed as this is the closest atmospheric nitrogen monitoring site to Exmoor National Park.

2.3 Pheasant/gamebird rearing

It is expected that pheasant/game shoots and associated game bird rearing activities are a special source in the area. A desk-based study and literature review was undertaken to understand the current available evidence for their likely contribution to air quality. Reporting was made available alongside Natural England data on Gamebird releases in and around (within 2km) Exmoor National Park, where releases occurred within 500m of an SAC. Natural England licences the release of gamebirds annually and is required to consider any likely significant effects on habitats sites (SAC and SPA). The evidence presented here will inform Natural England's consideration of future licence requests.

2.4 Other Data

A workshop was conducted with land managers, including representatives from the Exmoor Hill Farming Network (EHFN) and the Farming and Wildlife Advisory Group South West (FWAGSW), to gain an understanding of typical land management practices within the National Park. Additionally, a questionnaire was distributed during an EHFN meeting to gather further insights.

The information collected from these activities was used to develop a general overview of land management practices within Exmoor National Park. The objective was to compare this overview with the underlying assumptions of land management used in national air quality datasets, such as those available through the Air Pollution Information System (APIS). This comparison enabled the project team to assess the reliability, accuracy, and applicability of national-scale data and models to the specific context of Exmoor. Ultimately, this helped to understand the nuances of land management in Exmoor and critically evaluate any constraints and limitations of the data and evidence we are using.

2.5 Thresholds – Critical Level (CL_e) and Critical Load (CL_o)

There are two relevant thresholds associated with nitrogen pollution on Exmoor National Park: the critical level for atmospheric ammonia concentrations (NH₃-CL_e) and the critical load for nitrogen deposition (Ndep-CL_o). Critical levels, for vegetation, are defined as: “concentration, cumulative exposure or cumulative stomatal flux of atmospheric pollutants above which direct adverse effects on sensitive vegetation may occur according to present knowledge” (ICP Vegetation, 2017). Empirical critical loads are defined as: “a quantitative estimate of deposition of one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge” (Nilsson and Grennfelt, 1988). Critical loads and levels can be presented as ranges of values. This is to capture the sensitivity of all of the different species within each feature/habitat, as different species have different sensitivities



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to N. Generally, the critical load/level used for each habitat would be the lowest value for the most sensitive feature within the site.

As previously noted, the critical level for atmospheric NO_x concentrations is not regarded as an issue in Exmoor National Park. This is because elevated concentrations are typically associated with combustion sources, including sites of heavy industry and major highways, which are generally absent in the vicinity of Exmoor, noting the exception of Minehead and South Molton (see Section 3.1). However, as outlined above, NO_x emissions from such sources in the wider region can contribute to total N deposition; NO_x is therefore implicitly included when modelling priority habitat exposure to N deposition.

As shown by the definitions, the drive to set these values came about from observations that certain amounts of nitrogen, whether in the air or once it is processed through into the soil of an ecosystem, can be directly or indirectly harmful to vegetation and the wider habitat. For nitrogen deposition, harm is perceived to arise from its acidifying and eutrophying impacts which negatively affect conservation values, while for NH₃ concentrations, there can be direct phytotoxicity once compounds interact within or at the surface of the leaf, sometimes leading to mortality.

NH₃-CLE and Ndep-CLO values are regularly reviewed, with the most recent review for ammonia occurring in 2022 (Franzaring and Kössler, 2023) and that for empirical critical loads in Bobbink et al. (2022).

Importantly, critical levels typically refer to vegetation at a general level and are (generally) a single threshold value, although one that can differ depending on whether lichens and bryophytes are an important part of a given priority habitat area. **For Exmoor National park, there are two annual critical levels to note: the CLe for lichens and bryophytes which has been set to 1 µg NH₃ m⁻³, and the CLe for higher plants which is 3 µg NH₃ m⁻³ for (with an uncertainty of 2-4 µg m⁻³).**

- 2.5.1 Empirical critical loads are defined at the habitat level and are provided as a range, occasionally with guidance as to which part of the range may apply in particular circumstances. In some cases, critical loads can be modified and/or interpreted according to environmental context, for instance dry heaths and Northern Atlantic wet heaths should have a critical load used towards the high end of the range under high intensity management, and the low end of the range under low intensity management.

Table 1 provides the critical load ranges for a set of habitats expected within Exmoor National Park, especially the SACs and SSSIs.



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Table 1: Critical load ranges for a set of habitats expected within Exmoor National Park

Habitat Class	Critical Load minimum	Critical Load maximum
Acidophilous Quercus forest	10	15
Atlantic upper-mid & mid-low salt marshes	10	20
Broadleaved deciduous woodland	10	15
Carpinus and Quercus mesic deciduous forest	15	20
Dry heaths	5	15
Fagus forest on non-acid and acid soils	10	15
Inland sanddrift and dune with siliceous grassland	5	15
Low and medium altitude hay meadows	10	20
Moist or wet mesotrophic to eutrophic hay meadow	15	25
Non-mediterranean dry acid and neutral closed grassland	6	10
Northern Atlantic wet heath: 'L' Erica tetralix dominated wet heath (lowland)	5	15
Raised and blanket bogs	5	10
Rich fens	15	25
Valley mires, poor fens and transition mires	5	15

The values for critical loads and critical levels are set by international panels of experts, based on observations from experiments and/or surveys. Thresholds are set as a single value or a ranges. In the most recent revision of empirical critical loads for N, survey studies across spatial, temporal and/or spatio-temporal pollution gradients were considered. A range is given for each critical load, reflecting uncertainties in the data on deposition and on ecosystem responses, as well as different sensitivities of different examples of the habitat. An overall uncertainty is also indicated for the range given for each habitat (Bobbink et al. 2022).

There are various methods that can be used to assess the current degree of threat or impact from nitrogen deposition to a priority habitat or protected site (like those within the Exmoor National Park). For example, the critical loads set within Bobbink et al. (2022), which are assigned to UK priority habitat types and protected sites



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through the Air Pollution Information System³ and UK Critical Loads Trends Report⁴. Or the Nitrogen Decision Framework method, published by Jones et al. in 2016, which takes greater account of the uncertainties but would need to use the revised 2022 critical loads and updated EUNIS classifications.

The National Park Authority should be aware, that even with reductions in pollutant pressure, recovery of ecosystems may not follow immediately due to lag effects (i.e. delays in response, particularly true of biotic elements) or even hysteresis (where the same environmental conditions have different ecosystem properties associated with them due to internal system feedbacks that prevent return to the previous (desired) state (see e.g. Maes et al. (2024)). There is also evidence that for woody ecosystems, policy-relevant thresholds may not lead to recovery (e.g. Dirnböck et al. 2018). Efforts are ongoing to better characterise expected recovery trajectories from reduced N deposition (e.g. Perring et al. 2024) and/or reduced exposure to NH₃ concentrations.

Understanding of ecosystem recovery from air pollution impacts is still in its early stages. However, such information is increasingly needed by policymakers and land managers to demonstrate progress toward emission reduction commitments and broader legislative goals, including the maintenance and restoration of biodiversity. The [Air Pollution Recovery Indicators \(APRI\) project](#) - currently underway on behalf of Natural England—aims to address this need. It builds on a 2023/2024 scoping study commissioned by the Joint Nature Conservation Committee. Although limited data are available from the cessation of long-term nitrogen fertilisation experiments, these sites present a valuable opportunity to revisit and assess recovery processes in a robust and scientifically sound manner.

As the project is ongoing, it is not yet possible to recommend definitive sets of recovery indicators for specific habitats. Nevertheless, the preliminary findings highlight key data gaps and point to the need for targeted research. The APRI project is developing recommendations for field experiments and survey approaches that will support the eventual creation of practical guidance for identifying and tracking air pollution recovery indicators.

³ www.apis.ac.uk

⁴ https://uk-air.defra.gov.uk/assets/documents/reports/cat05/2411141006_Air_Pollution_Trends_Report_2024.pdf



3. Exmoor National Park

This section presents and interprets the evidence listed above for overall trends and patterns for Exmoor National Park as a whole. SACs and SSSIs cover a substantial area of the national park (Figure 1 and Figure 2), therefore understanding the spatial patterns across the whole area are useful to assess the nitrogen threats. Detailed site profiles for each protected site are in Section 4 of this report.

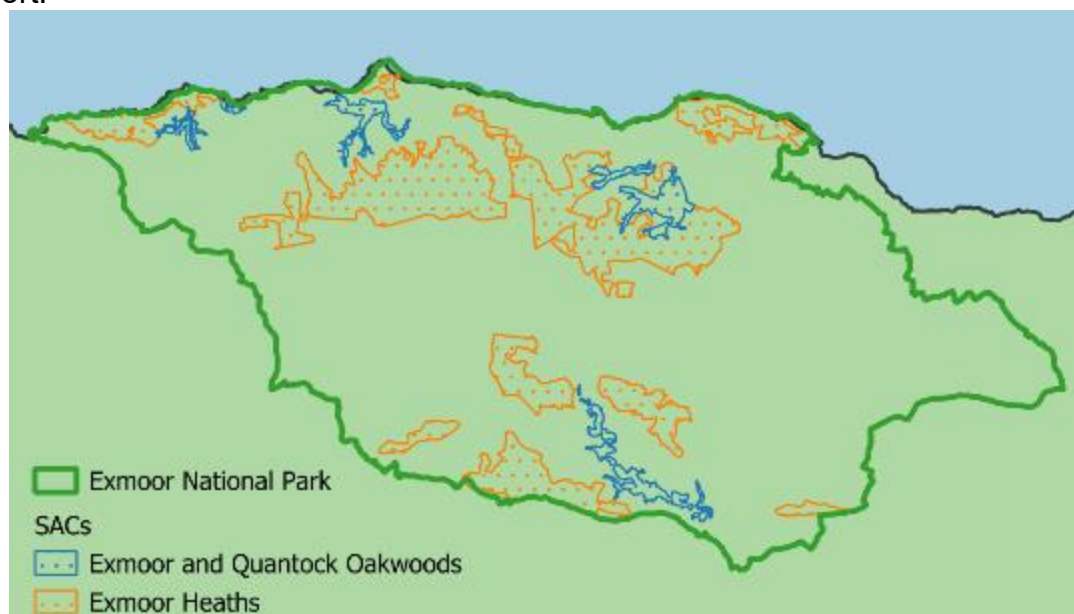


Figure 1: Location of SACs in Exmoor National Park



Figure 2: Location of SSSIs in Exmoor National Park

3.1 Emission Sources

Figure 3 illustrates the relative strength of emission sources from the main forms of atmospheric nitrogen pollution. **The data indicate that nitrogen pollution from agricultural ammonia constitutes the largest emission source in and around Exmoor National Park. Additionally, there are notable hotspots of non-agricultural NO₂ emissions, notably in Minehead and South Molton, which are primarily from combustion sources such as roads and industry.** It is important to note that these data are represented in 1x1 km grid cells, which tend to smooth out high emission peaks within each grid square across the entire area. Hotspot emission sources may exist and these could be masked by averaging emissions across the wider area.

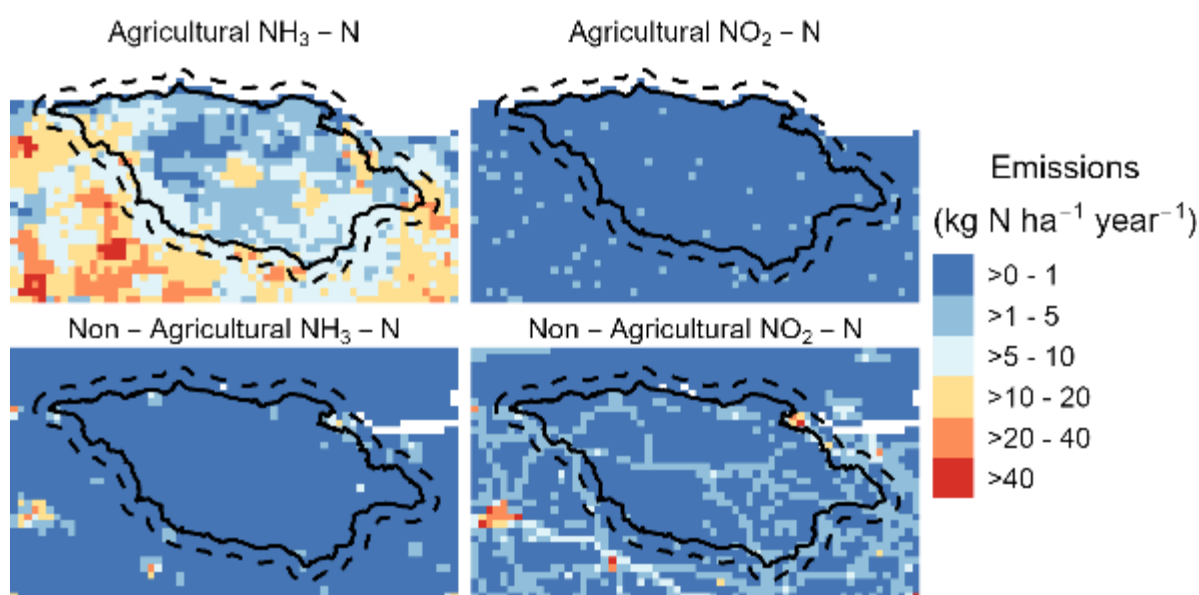


Figure 3: NAEI gridded 2021 agricultural and non-agricultural emission estimates (1km x 1km grid resolution). Dashed line indicates 2km buffer around National Park boundary.

The mean estimated agricultural ammonia (NH₃) emission within Exmoor National Park is 6.1 kg NH₃ ha⁻¹, with emissions ranging from 0.2 to 33.8 kg NH₃ ha⁻¹ (Figure 4). The gridded data shows a number of hotspot areas of elevated emissions within the National Park. A small central region near Exford which is estimated to have emissions between 15 and 20 kg NH₃ ha⁻¹ and is situated within approximately 2 km of the North Exmoor SSSI. Aerial imagery analysis suggests this could be due to a higher density of farm buildings and potential dairy or beef farming (Figure 84). Another area of interest lies to the southeast of Minehead, near Dunster Park and Heathlands SSSI, though this location is not in close proximity to any Special Areas of Conservation (SACs). Additionally, a higher emission region is observed to the west of Challacombe, near the boundary of the North Exmoor SSSI and in the vicinity of the central Exmoor heathlands.

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To maintain data confidentiality, spatial aggregation of agricultural census data was applied, requiring a minimum threshold of five farms per aggregated area. Therefore, the emission estimates and identified hotspots may differ from the underlying census data presented in Figure 4. The NAEI agricultural emission estimates are derived using average Emission Factors specific to robust farm type. A robust farm type is a classification of farms based on their main income source e.g. a farm with mainly poultry would be classed as specialist poultry⁵. Some of these farms are further classified depending on whether the farmland is favourable to agriculture or not. For Exmoor, it is likely that the majority of farms would be classed as LFA (less favourable area) grazing livestock and therefore have lower emission factors in the national modelling.

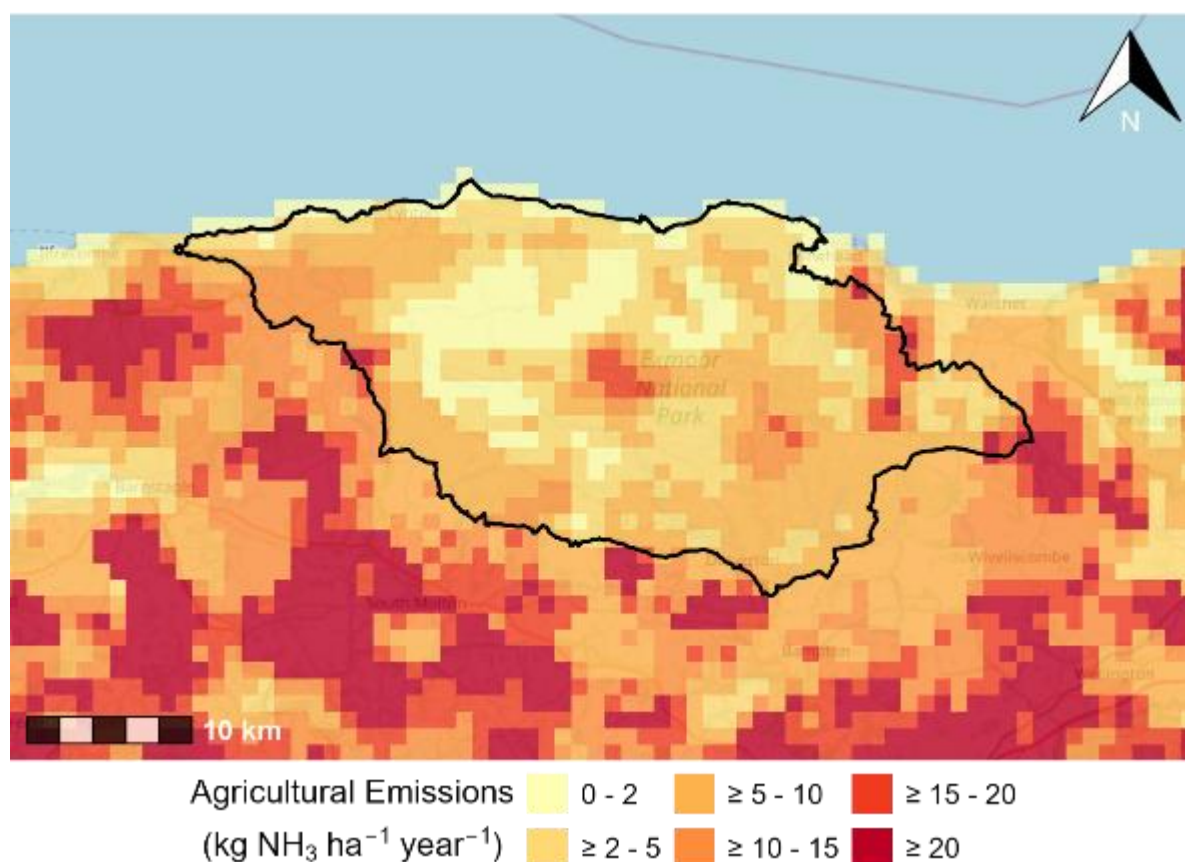


Figure 4: NAEI gridded 2021 agricultural emission estimates (1km x 1km grid resolution)

Agricultural census data for 2020 was used to identify the dominant emission sources in each grid cell (Figure 5). Mixed agriculture was prevalent in most of the central and northern regions of the national park, as well as in some southern areas. The model data shows that beef as a dominant emissions source towards the southeast and in a small area towards the centre of the park. Along the inland boundary, model data shows mainly mixed cattle to the east and small areas of

⁵ <https://assets.publishing.service.gov.uk/media/641073c8e90e076cd09acda9/fbs-uk-farmclassification-2014-14mar23.pdf>

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dairy as dominant sources. This data is derived from the agricultural census, which requires aggregation when fewer than five farms are present in a grid cell. This spatial aggregation can result in some variation between the modelled data and the census data. Consequently, when examining the data on a smaller scale and incorporating local knowledge, some discrepancies may be observed.

Figure 6 shows the agricultural emission density for concentric overlapping buffer zones outside of Exmoor national park. Emission data from the agricultural census are aggregated by emission source and divided by the buffer zone area to show the relative strength and type of emission source from the immediate vicinity of the national park boundary. The data clearly shows the strong influence of Beef and Dairy, contributing over half of the emissions per hectare within 1km of the National park boundary and throughout all of the buffer zone areas. Mineral Fertiliser is a relatively small proportion, as are pigs and poultry.

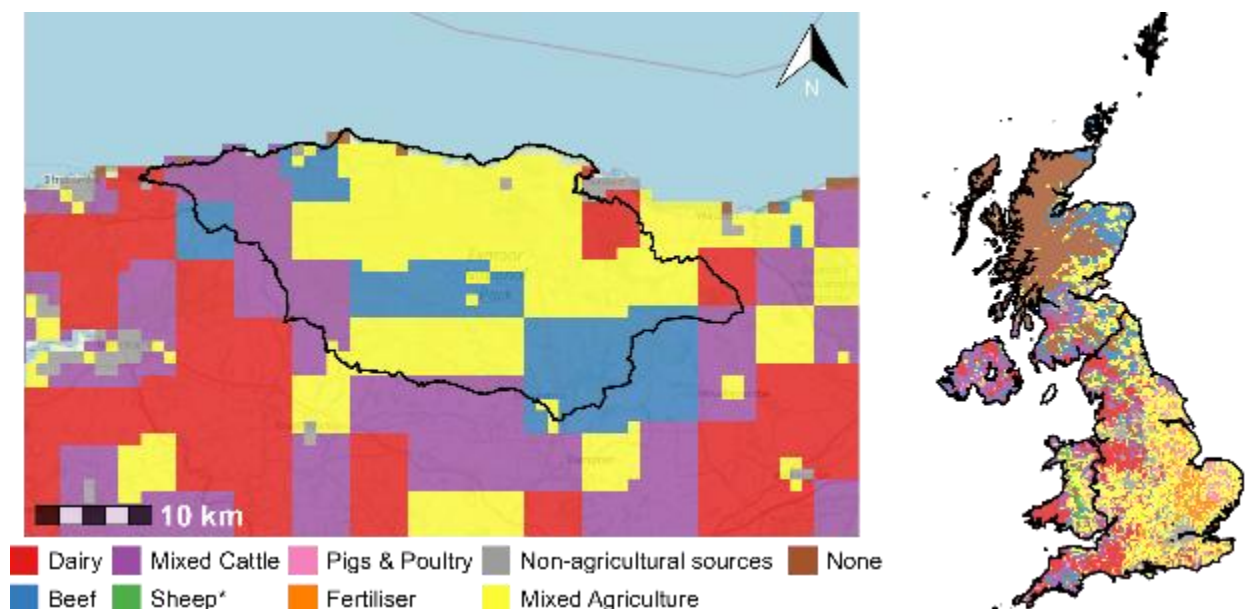


Figure 5: Non-disclosive dominant NH₃ emission sources (2020) across Exmoor National Park and UK.

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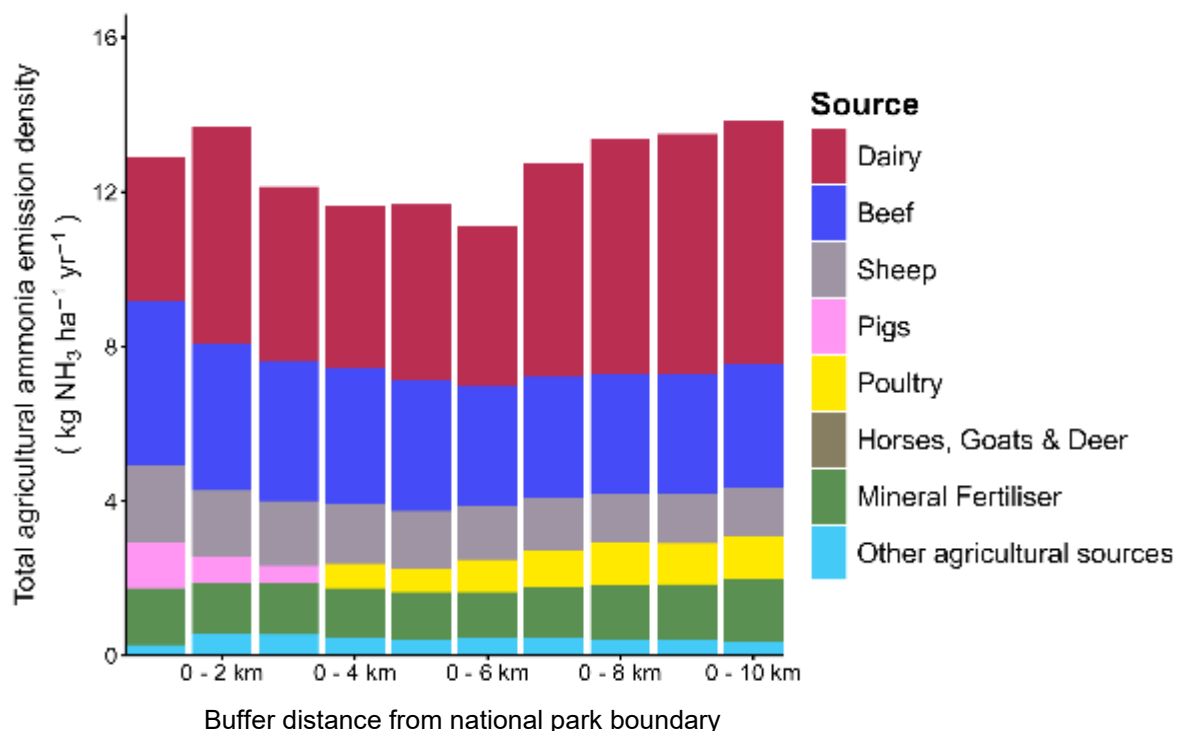


Figure 6: Agricultural emission density estimates for Exmoor National Park for concentric overlapping buffer zones outside of the National Park. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

To get a picture of the regional differences within Exmoor National Park and better understand the emission sources and strengths of emissions in its surrounding buffer zones, Agricultural census data were summarised by wind quadrants (Figure 7 and Figure 8). Wind quadrants were derived based on 8 wind directions (N, NE, E, SE, S, SW, W and NW) from the centre point of the National Park. Within Exmoor National Park, emission density was highest in the west, southwest and southeast, although these levels remained relatively low, reaching up to $10 \text{ kg NH}_3/\text{ha}$. The relative strength of emission sources varied across quadrants, with beef and sheep farming being predominant in all regions except the north. Additionally, the eastern region indicates the presence of poultry farming, which is not observed in other areas.

It is important to note that, to protect the confidentiality of farm data from the agricultural census, exact farm locations are not disclosed. Consequently, there may be some discrepancies when comparing this data to local knowledge. This may be the case for the western region, where the data suggests the presence of pig and dairy farms within the park boundary, which may actually be located just outside the National Park (but could still influence protected sites in this area).



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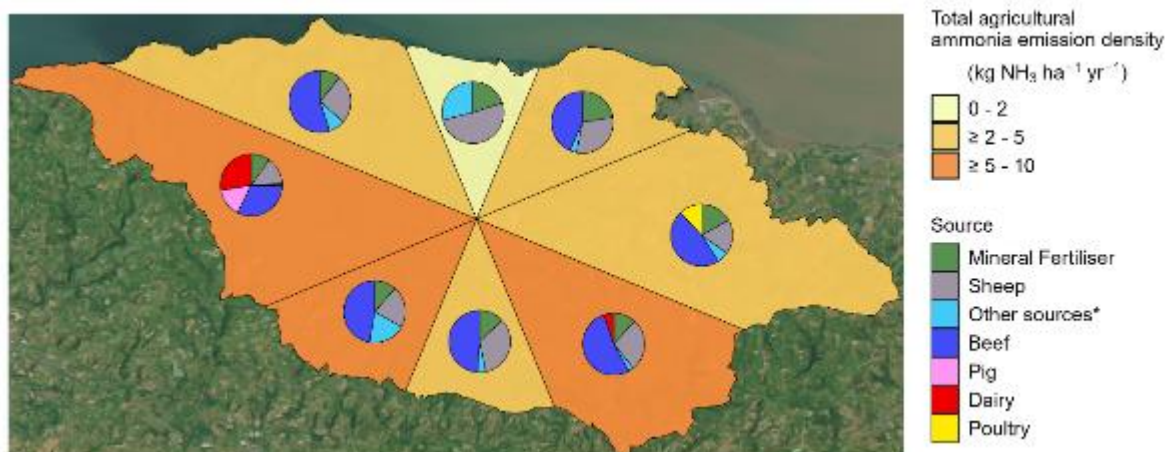


Figure 7: Agricultural emissions density and source for wind quadrants inside of Exmoor National Park. *The category 'other sources' refers to any other agricultural emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

In the buffer zones (Figure 8) area there are high density agricultural emissions within 1 and 2 km of the national park boundary to the west, with emissions of over 20 kg NH₃/ha. Emission source pie charts suggest a high prevalence of dairy emissions in this area. No other regions within 2km of the park boundary go above 15 kg NH₃/ha. There are slightly higher emission sources in the 1-2km buffer area to the south of the national park, with this potentially coming from beef and poultry emission sources. To the west, where emission density gets higher a few km from the boundary, dairy is a notable emission source.



Figure 8: Agricultural Emission Density and source for wind quadrant of buffer zones outside of Exmoor National Park. *The category 'other sources' refers to any other agricultural emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

3.2 Ammonia Concentrations

The EMEP4UK gridded model data for ammonia concentrations across Exmoor estimates slightly higher concentrations above $1\mu\text{g}/\text{m}^3$ along the inland boundary of the National Park (Figure 9). There is a larger area of above $1\mu\text{g}/\text{m}^3$ ammonia concentrations to the southeast of the National Park. This covers the southern areas of Exmoor Heaths SAC, as well as Barle Valley SSSI (which is a component of Exmoor and Quantock Oakwoods SAC). Annual mean values above $1\mu\text{g}/\text{m}^3$ are of concern for any ecosystems where lichens and bryophytes are a key component. Values above $3\mu\text{g}/\text{m}^3$ would be a concern for higher plants more generally, and such concentrations are absent from within the park boundary. Outside of the national park boundary, there is an elevated concentration hotspot to the southwest and, on the boundary of the mapped area, to the southeast. Neither of these hotspots exceed the higher plant critical level.

The central and coastal regions of Exmoor are estimated to be below $1\mu\text{g}/\text{m}^3$. This encompasses the larger central upland heaths part of North Exmoor SSSI and the central and eastern areas of Exmoor Coastal Heaths SSSI (both SSSIs are part of the Exmoor Heaths SAC).

The modelled data gives us a good idea of spatial patterns and the magnitude of concentrations, but any specific emission sources would be "averaged out" across the whole model grid squares. For example, a (theoretical) livestock house right next to a designated site boundary would be a local emission and concentration "hot spot" and affect vegetation and potentially fauna at the designated site, but this would be averaged out across the model grid square and not reflected in the figure below.

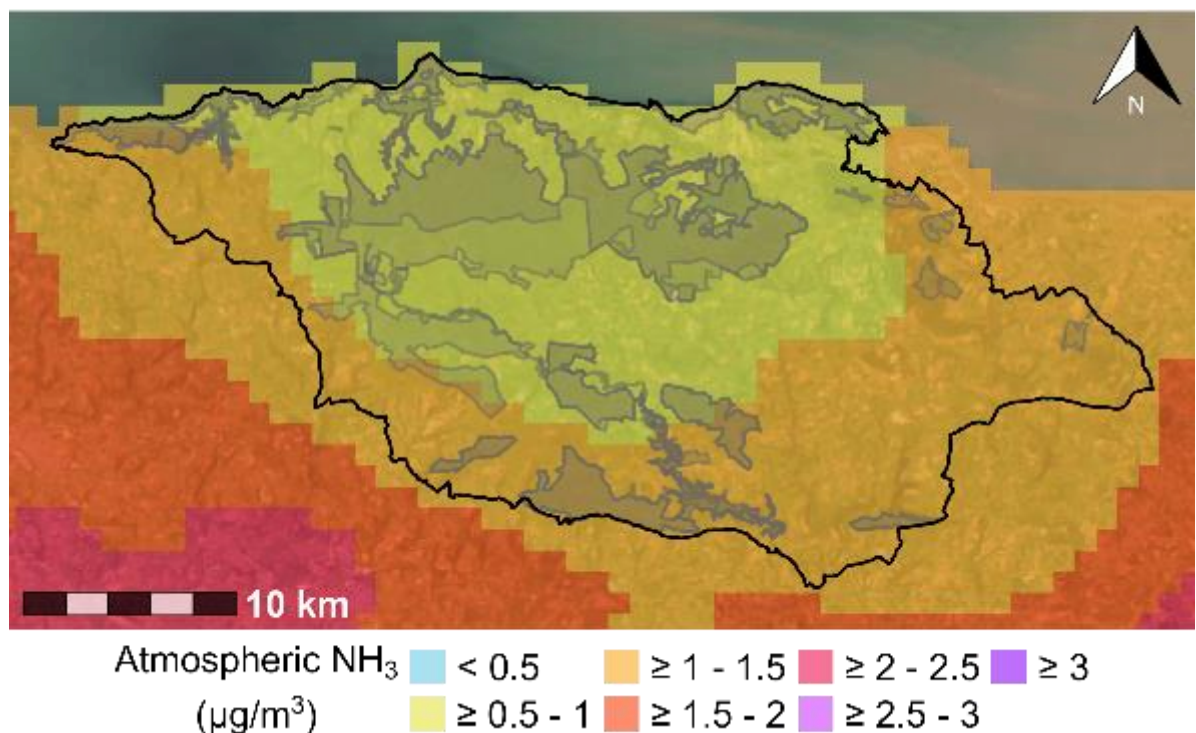


Figure 9: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Exmoor National Park. Protected Site areas are outlined in grey.

Braunton Burrows SAC is a coastal site approximately 24km west of the western boundary of Exmoor National Park and is a coastal dune site. Monthly NH_3 concentrations measured with the passive ALPHA® sampler method for the period May 2023 to July 2024 are shown in Figure 10. Monthly concentrations range between 0.3 and 1.6 $\mu\text{g NH}_3 \text{ m}^{-3}$. Ammonia concentrations at Braunton Burrows SAC were lower than the 1 $\mu\text{g}/\text{m}^3$ critical level for the majority of months between May 2023 and Dec 2024, with concentrations rising above 1 $\mu\text{g}/\text{m}^3$ in summer and September of 2023 and May 2024. Elevated concentrations are typically associated with spring (between March and May) and may therefore be linked to the spreading of manures/slurries on nearby fields during the land-spreading season. Concentrations are typically lower during the winter months (< 1 $\mu\text{g NH}_3 \text{ m}^{-3}$) during periods of reduced agricultural activity (and the closed season for spreading of manures).

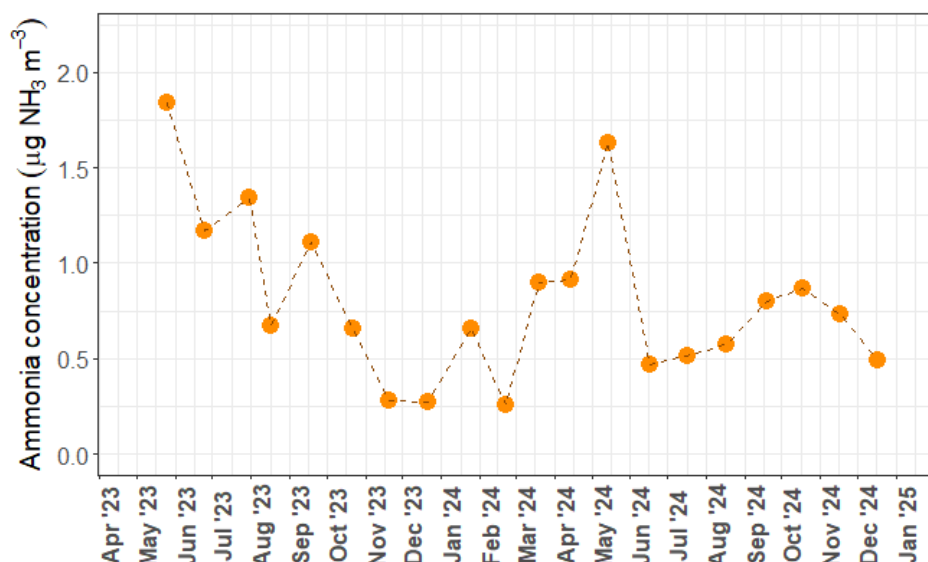


Figure 10: National Ammonia Monitoring Network ammonia concentration data for Braunton Burrows SAC

3.3 Nitrogen Deposition

The CBED model calculates N deposition separately for low growing semi-natural vegetation, such as heathlands and woodland vegetation (Figure 11). This is to account for the extra N deposition onto the rougher woodland surface. The pattern of N deposition is similar pattern for both vegetation types across Exmoor, with higher rates for deposition in the southwestern area of the National park, which gradually reduces further towards the coast.

The revision to empirical critical loads for nitrogen (Bobbink et al. 2022) indicates that **the whole of Exmoor can be considered above the lower limit of the**

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critical load for heathlands (which is 5-15 kg N/ha/yr). Deposition of N onto forests generally exceeds the lower limits of the critical load ranges for these habitats (set to 10-15 kg N ha⁻¹ yr⁻¹ for broadleaved temperate deciduous forest, *Fagus* (beech) forest on acid or non-acid soils, and acidophilous *Quercus* (oak) forest; and set to 15-20 kg N ha⁻¹ yr⁻¹ for mesic hornbeam (*Carpinus*)-oak (*Quercus*) forest), although exact locations of particularly n-sensitive habitat should be considered. **Overall, nearly all of the area of both open and wooded habitats within ENP receives more N deposition than the lower limit of the critical load range, indicating that harmful effects are likely to occur.**

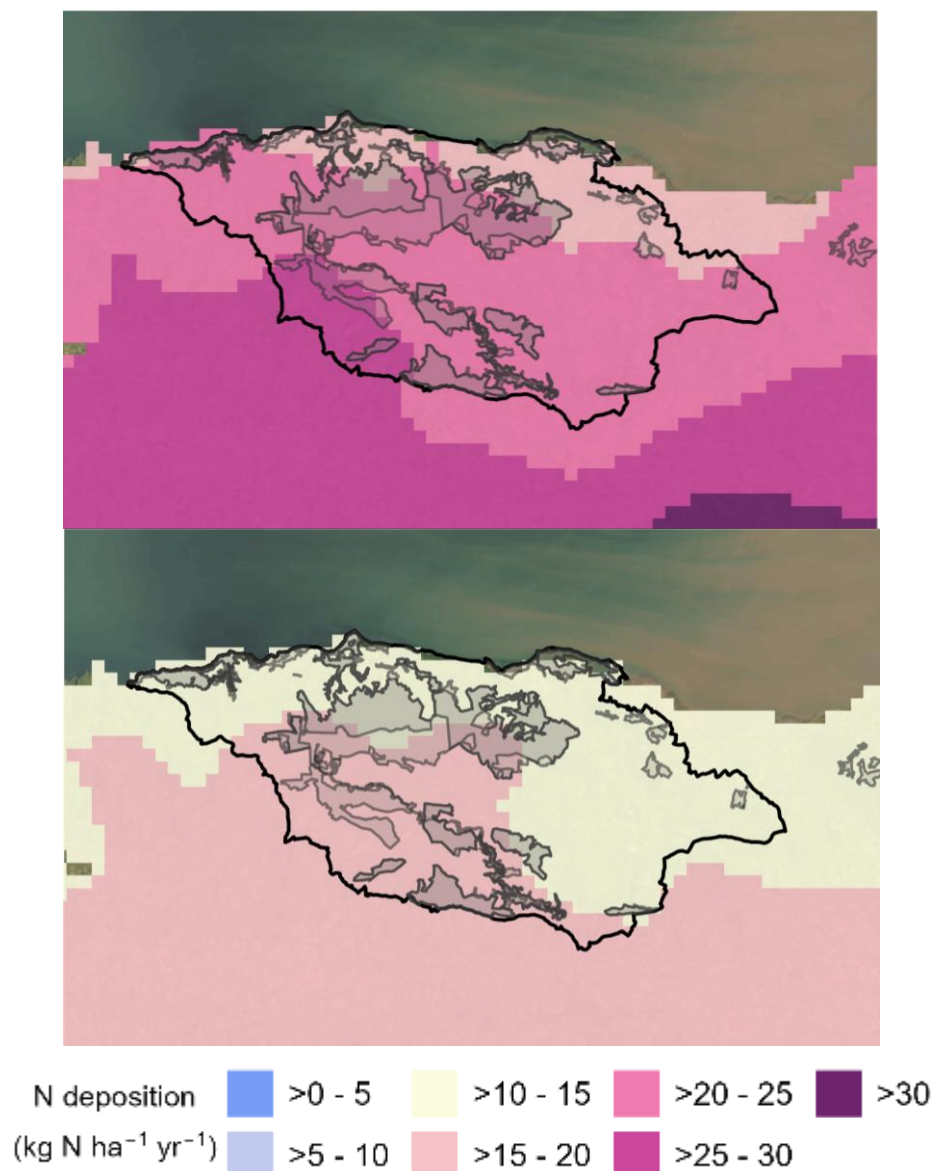


Figure 11; CBED mean modelled N deposition for 2019 – 2021 onto woodland (TOP) and low-lying vegetation (BOTTOM) features, for Exmoor National Park. Protected Sites shown in transparent grey.

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Source attribution data, as summarised in APIS, estimated the modelled source of deposited nitrogen pollution onto the Exmoor National Park. This is a combination of local, regional, UK and international sources. While the purpose of the AQMS is to focus on local sources, seeing the whole profile of N deposition onto the park is useful for determining how much can be addressed from local changes and how much will come from regional, national, and international changes and legislation to reduce nitrogen pollution.

For Exmoor as a whole, 45.7% of N deposition is estimated to come from emissions from Livestock in England, either within or outside the National Park. Due to the nature of how atmospheric nitrogen pollution behaves and reacts chemically, it can be assumed that a substantial proportion of this is likely to be from emission sources inside of or within a few kilometres of Exmoor. International shipping is the next largest source, at 9.7%, which is likely due to shipping lanes in the Bristol and English channels. Road transport from England is estimated to contribute 7.1% of nitrogen deposition on Exmoor.

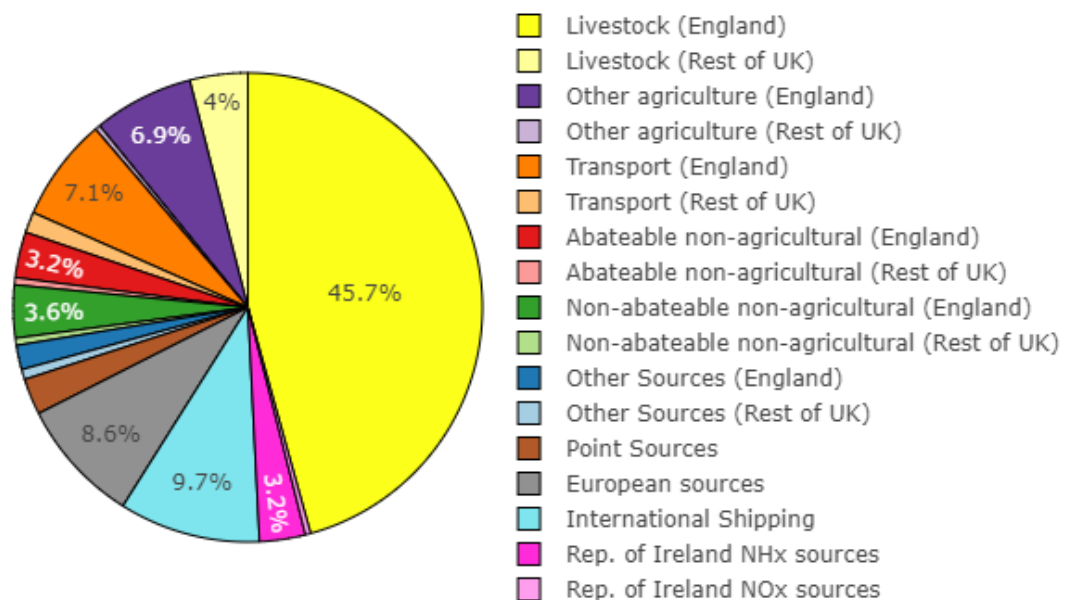


Figure 12: Source attribution (2018) for N Deposition onto whole of Exmoor National Park. Note: different sections of the sites may be subject to different sources of N deposition.

3.4 Gamebirds

Context and Summary

Understanding the ecological impacts of gamebird release was identified as a highly important and relevant topic for the UK policy and was identified as one of the top 100 questions of high policy relevance out of over 1000 questions posed to 28 different policy organisations by Sutherland et al. (2006). There have been



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concerns (e.g. Madden and Sage, 2020) that the increasing release intensity of non-native gamebirds (ring-necked pheasants (*Phasianus colchicus*), red partridge (*Alectoris rufa*)) recorded in the UK (Madden, 2021) may have air quality implications and subsequent impacts on biodiversity and ecosystems, including SSSIs and SACs. This concern is especially acute where gamebird release may occur on or near such protected habitat features. Given the presence of gamebird release in and around Exmoor National Park, the Park Authority wished to understand the extent to which gamebird release needs to be considered within the wider Air Quality Strategy.

Typically, the concerns in relation to impacts on air quality suggest there will be a build-up of nutrients in release pens from gamebird faeces that then translates, amongst other effects, into elevated ammonia (NH₃) concentrations, and potential localised elevated dry N deposition (Sage et al. 2020; Madden and Sage 2020; Sage 2018). As gamebirds disperse through the season, although for pheasants rarely beyond 1 km from a release site (Madden and Sage 2020), concentrated faecal deposition declines and thus air quality impacts likely lessen, beyond possible 'legacy' emissions related to still-elevated nutrient levels, prior to pens being used the following season. On occasion, gamebirds can disperse but congregate in certain areas (e.g. woodland rides) which may lead to localised emission sources beyond release pens; this potential source remains unaccounted for.

Here, we assess the evidence from a UK context that pertains to air quality impacts from gamebird release in and around release pen sites. We find that evidence directly linking gamebird releases to air quality impacts are currently understudied but do suggest that concerns around impacts on certain groups (e.g. N-sensitive lichens and bryophytes) is warranted. However, there may be alternative explanations for the community changes observed (e.g. arboreal foraging, as recorded for both red-legged partridge and ring-necked pheasant (Li et al., 2023)), and this highlights the urgent need to directly test emissions profiles in and around release pens in a robust manner and accounting for other potential drivers of ecological effects. A recent exercise in understanding stakeholder perceptions included multiple pathways through which gamebirds can have negative or positive ecological impacts but overlooked an air quality impact pathway, and nor was this raised in answers to a free text question available to respondents (Minter et al. 2024). This emphasizes the need to characterise the gamebird release - air quality pathway and, where necessary, raise awareness as to its impact.

Data on actual gamebird release numbers and locations is highly uncertain (Madden 2021) even with FOI data provided by Natural England (Figure 13 and Table 2) this is only over 4 years and within a limited area (releases within 500m of SAC). Thus, assessing the effects on air quality of gamebird releases on specific SSSIs and SACs, or the Park as a whole, is beyond the scope of this short desk-based assessment. We make recommendations as to how to tackle this uncertainty in future work, including that it would be helpful to gather and assess data across the whole national park on release pen size and location, gamebird densities, and



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species composition, given the different life history/habitat use profiles and emission intensities of red legged partridge and ring-necked pheasant (e.g. Tomlinson et al. 2017) and likely impacts of release density on air quality (e.g. Mason et al. 2020; Madden and Sage 2020; Sage et al. 2020).

We emphasize that any recommendations relating to gamebird releases and air quality need to understand the wider socio-ecological context of gamebird releases, including taking account of perceptions of land managers (Minter et al. 2024). Considering the positive impacts of economic opportunity and social wellbeing with the negatives of human health implications - from lead ammunition, gamebirds as potential disease vectors and vehicle and aviation collisions (Mason et al. 2020).

Recent reviews by the RSPB and Natural England (Mason et al. 2020 and Madden and Sage 2020, respectively) highlight there may be some positive ecological impacts for biodiversity in areas surrounding gamebird releases. However positive effects are generally considered to only come from secondary or associated effects from type of management adopted by gamebird estates. Most direct ecological impacts of gamebird releases in these reviews were considered negative. Direct ecological impacts in and around the release pen site (e.g. through ground disturbance, foraging, roosting in trees), together with management impacts such as more open woodland canopies in release woodlands compared to controls, can also lead to ecosystem community change that could otherwise be attributed to air quality impacts, reinforcing the need to disentangle air quality effects in a robust manner. Another recent paper (Blackburn and Gaston, 2021) highlights the sheer biomass associated with gamebird release and its unusual seasonal profile in the context of the intra-annual cycle of native British bird biomass. Blackburn and Gaston (2021) estimated that around a quarter of annual British bird biomass is contributed by pheasant and red-legged partridge and at their peak in August they represent about half of all wild bird biomass in Britain, which will likely have important ecological implications. One such implication may be ammonia emissions associated with the concentration of so many birds in relatively confined release areas. Given the scale and density of releases year on year, it is plausible that elevated NH_3 concentrations and N deposition could occur locally, potentially altering soil chemistry, vegetation communities, and contributing to wider nitrogen enrichment effects. Further work may be needed to quantify and assess the ammonia and nitrogen related impacts of this annual pulse of biomass.

Thus, and hypothetically speaking, if recommendations were made to cease gamebird release on and around certain protected areas, there may be broader implications for habitat condition in the surrounding landscape. This would depend on, for example, whether land managers engage with schemes such as the Environmental Land Management Schemes (ELMS). While management for gamebirds can contribute to certain habitat features, is also associated with documented negative ecological impacts, which must be weighed in any such assessment. At the same time, air quality impacts associated with gamebird release, but not a direct impact from the release pen site, may also need considering via counterfactual analyses. For instance, how do actions around



gamebird release (e.g. growing of maize and other seed crops, travel to and from shooting locations) contribute to emissions profiles? How would emission profiles and thus air quality impacts on designated sites alter following successful implementation of any specific air quality strategy recommendations around gamebird release i.e. what activities, if any, would replace those restricted, if any, by any air quality strategy? More broadly, and as identified by Blackburn and Gaston (2021), what would be the response of community assemblages in the absence of these birds and the environmental manipulations their presence entails? Answering such questions is beyond the scope of this part of the report but would be a valuable next step in understanding likely impacts in the context of changing environmental conditions more generally. It should be considered that answering such questions can be difficult: according to Blackburn and Gaston (2021), and given the widespread abundant nature of non-native galliforms (heavy-bodied ground-feeding birds like pheasants and chickens), local scale responses are dependent on their macroecological context and so small-scale short-term gamebird exclusion experiments in a wider landscape where they are an abundant species will not quantify their effect. As birds like gamebirds are relatively common, studies that would look at what happens if gamebirds are removed from a small area might not be able to reliably quantify the effect of the removed gamebirds. This is because of the large number of other gamebirds nearby that could still be having an effect on the small area gamebirds were removed from.

Approach

To understand the potential impacts of gamebird release on air quality we first carried out a literature review at the UK-level, building on three recent authoritative reviews of ecological impacts, all published in 2020 (Sage et al. 2020; Mason et al. 2020; Madden and Sage 2020). To assess whether additional research had been published since these reviews we searched Web of Science, using two search strings, with an additional search in GoogleScholar, notwithstanding difficulties in ensuring repeatability with the latter.

In Web of Science, we first searched using the following topic terms across all years, in case any air quality specific papers had been missed in the earlier reviews (Search A):

(TS = "Phasianus colchicus" OR pheasant* OR "Alectoris rufa" OR "red-legged partridge*" OR "gamebird*" OR "game bird*") AND (TS = ammonia OR NH₃ OR nitr* OR "air quality") **Search A**

We assessed the titles and abstracts returned from this search ($n = 42$). None of the papers were relevant. Subsequently, and given that Search A failed to find any relevant papers, we used the following topic terms, for 2020 to 2024 only (Search B):

(TS = "Phasianus colchicus" OR pheasant* OR "Alectoris rufa" OR "red-legged partridge*") **Search B**



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This search returned 504 papers and having screened titles and abstracts with appropriate screening criteria (e.g. excluding based on geographical context, subject matter), we downloaded 7 papers for further inspection.

We also briefly reviewed records found through a GoogleScholar search, restricting ourselves to a manageable number of references to inspect, by using the following search terms bounding publication date between 2020 to 2024 only (Search C):

("Phasianus colchicus" OR pheasant* OR "Alectoris rufa" OR "red-legged partridge*") AND ("ammonia" OR NH₃ OR nitr* OR "air quality") **Search C**

In addition to any new literature revealed by these approaches, we considered the findings from Madden and Sage (2020), Sage et al. (2020), and Mason et al. (2020) together with any literature recommended by the wider co-authorship team, when assessing the evidence for air quality effects, inspecting original source references where possible.

Findings

The Gamebird Release Process: The Potential for an Air Quality Impact

Although no specific studies were identified that directly quantify the effects of gamebirds on air quality, pre-existing reviews and literature published since then provides a mechanism by which gamebirds - especially at high densities in release pens - can influence air quality. In brief, the literature, which generally concerns pheasants with some reference to red-legged partridge, suggests that faecal deposition, associated with supplemental feeding as well as existing food sources within and around release pens, will lead to elevated nutrient levels, specifically of nitrogen (e.g. Madden and Sage 2020; Mason et al. 2020). These elevated levels can affect soil microbial processing while ammonia (NH₃) is released from the faecal deposits directly. As density of gamebirds increases, it is expected that such gaseous release will also increase, and thus lead to elevated atmospheric concentrations of nitrogenous gases in, and downwind of, release pens. This provides a basis for caution around N sensitive and protected sites and highlights a need for further investigation, particularly given the scale and concentration of releases in some landscapes.

Otherwise, nitrogenous compounds can react in the atmosphere and, over time, be deposited through dry and wet N deposition. Typically, these depositional pathways ameliorate and integrate emission sources such that it would become difficult to attribute a unit of N deposition on a particular protected habitat to a particular gamebird release site (although see source attribution diagrams for Exmoor as a whole in Figure 12 or by site in the Site Profiles in Section 0). Depending on climatological conditions, the relative importance of dry and wet (and occult – through fog and mist) deposition will vary; it might be expected that dry deposition will dominate in the immediate vicinity of release pens due to surficial vegetation deposition and subsequent throughfall, while wet deposition of atmospheric nitrogen will dominate as concentrations move through the atmosphere and get



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removed by rainfall. The relative importance of dry and wet deposition may also depend on the habitats concerned, with greater surface roughness of wooded habitats leading to increased amounts of deposition compared to low-growing vegetation.

Any nitrogen deposition that does enter an ecosystem can have deleterious impacts on sensitive habitats through both acidifying and eutrophying pathways e.g. at very high levels leading to aluminium (Al) toxicity from pH changes, nitrate leaching to ground- and surface waters, and increased dominance of particular elements of biodiversity (across trophic levels) as competitive plant species take advantage of the high nutrient availability, including homogenization of plant communities across space and time (as recently summarised in e.g. Perring et al. 2024).

The presence of gamebirds in release pens is for a relatively short period of time (for pheasants when they are 6 – 8 weeks old in July or August, until the beginning of the shooting season on 1st October) (noted in Hall, Sage, and Madden 2021). Towards the end of that period, gamebirds are encouraged to disperse through the landscape. For instance, pheasant-rearing pens are open to the woodland canopy allowing birds to forage away from the sites as they get older and develop a greater ability for flight (Hall, Sage, and Madden 2021). After dispersing, and with the beginning of the shooting season, the birds tend to get distributed throughout a landscape, although dispersal distances can be limited (e.g. typically less than 1 km (Turner (2007) and/or Beardsworth et al. (in prep) cited within Madden and Sage 2020)), with mortality occurring throughout the pen- and dispersal-periods, through multiple pathways (e.g. disease, shooting, predation, vehicle collision). While the intensity of faecal deposition (and associated ammonia release) is likely to decline at release pens as birds disperse, the large number of gamebirds in the wider environment may still contribute to diffuse nitrogenous emissions at a landscape scale. However, because these emissions are more spatially dispersed away from release pens, they are less likely to result in the high localised concentrations associated with toxic effects or acute ecological damage. Despite this, low-level nitrogen inputs from diffuse sources can still contribute to long-term ecological change, particularly in nitrogen-sensitive habitats, and the possibility of birds aggregating around feeding stations or sheltered areas may create localised hotspots even post-dispersal.

In the following, having introduced recent evidence for gamebird numbers and spatial distribution in the UK and more specific data for the Exmoor area, we investigate the extent to which gamebirds in release pens could potentially contribute to air quality impacts via directly toxic ammonia levels and/or atmospheric N pollution effects. Due to evidence limitations (e.g. observations of where gamebirds potentially congregate outside of release pens), we do not consider impacts from such gatherings. We make recommendations as to how a gamebird emission source could be included in N emissions and thence N deposition to protected sites, which will only be possible with an improved



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understanding of numbers and release sites in the vicinity of Exmoor, data that are not available for all releases over all of Exmoor at this time.

Gamebird Numbers and Distribution in the UK and Around Exmoor

Triangulating across 12 different approaches, Madden (2021) estimated releases of 31.5 million pheasants and 9.1 million red-legged partridges (Madden 2021). However, such figures differed substantially from official records and previous published estimates (e.g. Aebischer 2019) and highlight the critical uncertainties around gamebird releases. Of particular importance may be that Madden's (2021) most conservative estimate is more than twice that of UK Government Animal and Plant Health Agency (APHA) figures, which suggests there is a chronic underreporting of gamebird numbers and releases (as well as failure to comply with register regulations; cited as being less than 25% in Madden and Sage (2020)). This makes it very difficult to use official figures to understand likely gamebird impacts on protected sites from air quality or other impacts.

The pattern of release is variable across the country. In the absence of specific information, Madden and Sage (2020) estimate release patterns at a county level with relative intensity shown without an absolute scale (their Figure 2). Given the location of Exmoor (straddling Somerset and Devon), these figures show that Devon has slightly lower numbers of birds reported for release than Somerset (both partridge and pheasant at number of birds per km²), but similar number of shoots offering these quarry species per km². The number of pheasants reported as held for release in Somerset is amongst the highest in the country. Later maps (e.g. their Figures 4, 5 and 6) reinforce these patterns, including correcting for area of the country within which birds are held for release, with pheasants at higher density than partridge regardless of whether they are in Devon or Somerset (their Figures 4 and 5).

As noted above, it is likely these figures, especially when derived from APHA, are underestimates of releases. Furthermore, it is unknown whether this underestimation is consistent across counties or whether there are biases. For the purpose of the Exmoor AQMS, Natural England have supplied location data and size of game bird released inside of and within 2km of the national park boundary where they were within 500m buffer zone of a SAC and/or SPA. This data was available for 3 years from 2021 to 2023 (Figure 13). Total gamebird numbers under the same conditions were also released for 2021 to 2024 under a FOI requested by ENPA. As the data are relatively limited it remains unclear the extent to which sites are susceptible to ecological impacts from gamebird release, related to air quality or otherwise.



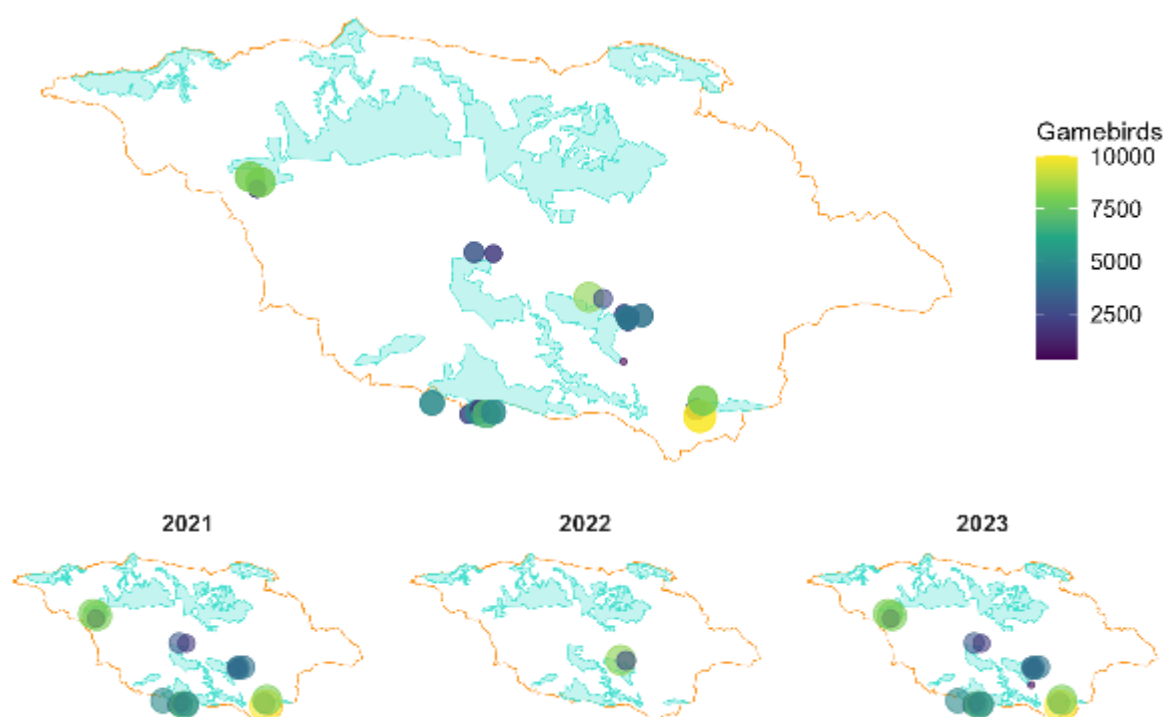


Figure 13: Gamebird releases in and around Exmoor National Park SACs. The size and colour of the point is proportional to the total number of birds released. TOP shows all gamebird release data between 2021 and 2023, BOTTOM shows individual years.

Table 2: Total number of pheasants and red-legged partridges released over the last three years in and around Exmoor National Park.

Year	Number of releases	Number of Pheasants	Number of red-legged Partridges	Total Number of Birds	Mean birds per release
2021	18	46,500	37,200	83,700	4,650
2022	2	10,800	0	10,800	5,400
2023	19	46,850	37,200	84,050	4,423
2024	21	77,250	32,200	109,450	5,212

Given the likelihood that air quality impacts (as well as other ecological effects) can depend on the size/intensity of releases (as discussed in Mason et al. 2020; Sage et al. 2020) it is important to consider whether gamebird releases on Exmoor are associated with large, commercial operations or smaller concerns. Feedback suggests that both small-scale and large commercial operations are present within the national park, with an apparent trend toward an increase in the larger

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commercial operations. Madden and Sage (2020) report that there are very skewed figures on release nationally: for instance, 6 out of 3307 pheasant shoots released more than 100,000 birds whereas the median release was 850 birds per shoot; partridge had a similar release profile with 2 out of 1323 shoots releasing more than 100,000 birds, but a median release size of 500 birds per shoot.

Using the figures on numbers of releases above, and the total number of birds, then you would get an overall and relatively consistent release intensity across years of 4,650 (2021), 5,400 (2022), 4,423 (2023) and 5,212 (2024) birds per release. Although these numbers could be impacted by bird flu and continued recovery from reductions during Covid-19, which may be lower than historic released within Exmoor. These numbers may also not reflect the reality on the ground i.e. it is unlikely that all releases across a year will release the same number of birds. Characterising this distribution will be important, as larger shoots may have disproportionately greater human activity and invest more financially and in terms of effort in land management activities such as game crop plantings, woodland management and supplementary feeding. They may also release birds at higher densities and/or be able to exercise more coordinated predator control over contiguous areas (Madden and Sage 2020). At the same time, evidence from 49 pens in central England, based on GPS co-ordinates of pens and manager-provided figures, showed that stocking densities were markedly higher than previous reports (ranging from 220 to 7310 birds per ha, with a mean of 2600 birds per ha). Recommendations are for stocking at less than 1000 (and less than 750 birds per ha for semi-natural ancient woodlands) (see Code of Good Shooting Practice, Game and Wildlife Countryside Trust: www.gwct.org.uk), although the evidence for these figures may relate to unusually small pens containing relatively few birds (Hall, Sage, and Madden 2021). These types of densities, regardless of release intensity, if replicated for Exmoor and surrounds, **emphasize the clear need to understand the numbers, spatial distribution and gamebird practices associated with gamebird release**. Given the potential for ecological impact, there is a case for Natural England to review regulation in this area, potentially by expanding licensing requirements or developing more robust guidance to ensure releases are environmentally sustainable.

Evidence for Gamebird Impacts on Air Quality: Emission Factors and Direct Effects

We did not find any published studies that directly quantify the impacts of gamebird releases on air quality in Exmoor with our literature search. However, this should not be taken to imply that such impacts do not exist—rather, it highlights a significant gap in current evidence. Although our literature search did not yield new studies on gamebird-related air quality impacts more broadly, literature published after 2020 (Mason et al. 2020; Madden and Sage 2020; Sage et al. 2020) are featured elsewhere in the Gamebird Section of this report. In the context of Exmoor, research by the Game and Wildlife Conservation Trust on the environmental impacts of pheasant release at local shooting estates has examined ecological effects but does not appear to include direct measurements of air quality or ammonia emissions. Therefore, it is appropriate to conclude that there is currently



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insufficient evidence to assess the scale of air quality impacts from gamebird releases on Exmoor, and that targeted research in this area is urgently needed. Notably, the fact that gamebirds contribute to ammonia emissions is highlighted by Tomlinson et al. (2017). In their update to the non-agricultural national atmospheric emissions inventory, they note how the category of pheasants was revised to 'gamebirds' in 2012 to account for the increasing number of red-legged partridge in the UK. This altered the emission source strength, weighting it to account for the smaller mass (on average) of a partridge to a pheasant. This led to an average emission factor for gamebirds of 0.017 kg NH₃-N per bird per year, a slight decrease from the 0.02 kg NH₃-N per bird per year for pheasants. Although this emission factor is more representative of the intensive rearing of gamebirds, being more akin to poultry sheds, rather than their release. Some data were available on the location and magnitude of gamebird rearing, but these only applied to sites within 500 metres of an SAC, limiting their broader applicability throughout the National Park. However, feedback from the evidence-gathering workshop suggested that gamebird rearing - particularly pheasant rearing - is prevalent in and around the National Park, with a high density of rearing pens located in woodlands. More work needs to be done to understand the emissions impacts of gamebird releases. It will also be important to understand how these per bird emissions over the course of a year scale to ammonia releases in and around release pens over the shorter timescale of their presence at such locations, and how these factors scale with mortality at, and subsequent to, release.

Here we briefly report on the indirect evidence that reports ecological impacts of gamebird releasing through their effects on air quality, including from within Exmoor (Sage, 2018). The reason these studies implicate air quality is because some species of mosses, liverworts and lichens tend to be particularly sensitive to atmospheric N and tend to only remain common in woodlands that are considered 'clean-air' (Mitchell et al. 2004). Madden and Sage (2020) then implicate unpublished data from the then Centre for Ecology and Hydrology as suggesting that some woodlands with large pheasant releases could have atmospheric N levels that are elevated to a level that could cause these sensitive species to decline.

Of most relevance to this report are recent findings from seven southern Exmoor estates, where liverwort species on tree trunks were two- to four-times more abundant in non-release areas of broadleaved woodlands compared to release-pen areas of such woodlands on the same shooting estate. In addition, mosses and liverworts were twice as diverse in the non-release woodlands as compared to those with pens (Sage, 2018). Most revealingly, the abundance and diversity of bryophytes and lichens on trees overall was not different between release pen plots and control plots in the same woodland but outside of the release pens, while moss diversity was about 25% lower on trees in both the release pen plots and nearby control plots (in the same woods), compared to the estate woods without release pens. However, there was no difference in lichen species diversity between plot types. Liverwort species diversity was between ~30 and 50% lower in the release pens and release pen wood controls, compared to the non-pen estate woods. For



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moss and lichen abundance overall, and abundance of common species such as *Isothecium myosuroides* and *Lecanactis abietina*, there was no difference between estate plot types, or between estate controls and other non-estate woods. Liverwort species as a group, and the common species *Frullania tamarisci*, were about 50% less common in pheasant release pens and in the control plots in the same woods, compared to the non-pen estate controls. *I. myosuroides* and *F. tamarisci* are relatively common species that are considered to be sensitive to atmospheric nitrogen enrichment. Given this, and the fact that impacts are seen in woodlands with gamebird release but beyond the release pens themselves, and not in non-release woodlands in the same landscape, Sage (2018) suggests that effects are potentially due to atmospheric N emissions.

Additional circumstantial evidence from elsewhere in the UK includes reports on concerns for lichens at Croxton Park SSSI in Leicestershire due to elevated ammonia and atmospheric N deposition levels (Smith 2014) while Bosanquet (2018) highlighted the potential legacy impacts of a 2014 gamebird release on the lichen flora of an upland acidic oak woodland at Allt-y-Gest SSSI in Wales following his survey in 2018. He identified relative degradation of the flora in the part of the woodland enclosed by a pheasant release pen, including the rare presence of an N-tolerant lichen species (*Physcia adscendens*) (recorded on three trees as scattered small thalli; more than 100 trees sampled overall) and another indicator of eutrophication on 8 trees: free algae (Bosanquet 2018). These records were not made on all recorded trees within the footprint of the release pen, but no signs were recorded on any trees outside the release pen footprint, despite actively seeking enrichment indicators elsewhere including alongside a road in the northwestern part of the wood (Bosanquet 2018). Rothero (2006), as cited in Madden and Sage (2020) reported changes to the soil composition and effects (reductions) on bryophytes at a SSSI in Scotland when the estate started to release red-legged partridges nearby; it is unclear whether this relates to air quality impacts or direct faecal deposition.

In all the cases detailed above, there were no measurements of actual air quality and other changes around the woodland release pens could explain the findings. For instance, Sage et al. (2018) highlight that the impacts on liverworts and mosses may relate to tree and shrub management, rather than air quality, but “we do not know this”; the effect of woodland management as explaining community changes was also noted more broadly in the recent evidence review (Madden and Sage 2020). As noted earlier, pheasants (and red legged partridge) can also forage and roost in trees (Li et al. 2023) and have an eclectic diet (Sage et al. 2020), so it may be that they can directly disturb mosses, liverworts and lichens through these processes rather than via air quality. This latter pathway may though be unlikely on those bryophytes and/or lichens that are found on vertical tree trunks. In other instances, it is hard to differentiate the effects of gamebird releases given high intensity livestock agriculture around the site (e.g. Croxton Park SSSI), although at least in Allt-y-Gest (Bosanquet 2018) the presence of other enrichment sources at the north-west side of the wood where the release pen was located was described as unlikely and there is a reasonably high confidence that unwanted ecological



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impacts can be ascribed to pheasant rearing. The potential impacts of landscape emission drivers are also minimised in the Exmoor example (Sage 2018) given the careful plot selection of (i) release pens, (ii) control plots in the same woodland outside of release pens, and (iii) plots in non-release woodlands in the same landscape.

Other Direct Gamebird Impacts on the Environment

Understanding the extent to which air quality is affecting ecological assemblages on protected sites requires disentangling its role from other direct (and associated) gamebird impacts on the environment. The raft of impacts, and underlying evidence, is summarised in recent reviews (e.g. Mason et al. 2020; Madden and Sage 2020) but includes responses across scales (Sage et al. 2020). Thus, at the local, release-pen scale, there are perceived negative effects on ground flora, soil and woodland ground invertebrates, where a density component and, in some cases, legacy effect from abandoned pens (Capstick, Sage, and Hoodless 2019) has been shown. Perceived positive effects include increased shrubs, butterflies and bees (Sage et al. 2020). At the patch scale, and besides impacts on woodland bryophytes and lichens on trees, direct impacts are reported for reptiles and include overshooting of wild partridges in areas where red-legged partridge are released, and, in the wider landscape, potential impacts on endo-parasites of gamebirds, spreading of gamebird disease to other wildlife components and direct effects on predator populations, all with some evidence for increasing impacts at increasing release densities (Sage et al. 2020).

Associated (Gamebird Management-Related) Impacts on the Environment

The management of woodland patches for pheasant release, and farmland for gamebirds more generally, appears to have ecological impacts across scales as well. For instance, at the local scale the presence of woodland rides in gamebird woods leading to habitat diversity; at the patch scale, songbird use of pheasant woods in winter (work by the Game Conservation Trust - Hoodless et al. (2006) - found more birds and a higher number of species in winter game woods, linking these differences primarily to structural characteristics created or maintained through game management practices), and the presence of small mammals; while at the landscape scale there can be increased woodland planting and retention (Sage et al. 2020). Generally, these impacts are understood as positive, notwithstanding alternative perceptions from a range of stakeholders (Minter et al. 2024). Perceived negative impacts at patch and landscape scales include effects of released pheasants on hedgerows, again, with a density component and the releasing and illegal killing of raptors and associations between game birds, ticks and *Borrelia* (Sage et al. 2020).



Recommendations

To further understand the potential for gamebird releases to affect air quality and thence to affect protected site status at Exmoor (or elsewhere) we make the following recommendations:

- Under the precautionary principle, avoid gamebird releasing activities in close proximity (typically sources within 1-2km of a protected site are considered in a Site Nitrogen Action Plan) to protected sites that derive their designation from the presence of sensitive lichen and/or bryophyte species especially where such activities may push ammonia concentrations / N deposition beyond critical levels / critical loads. See the Site Profiles section of this report for a full list of relevant sites on Exmoor, their qualifying features, relevant lichen and bryophyte important and associated critical levels and loads. Where pollutant levels are already high from other activities, adding pollution will further compromise the integrity of such sites, and it would be beneficial, from a conservation standpoint, to encourage pollution reduction efforts.
- To inform evidence-based policy, future work should measure atmospheric concentrations of ammonia (and, where possible, other N-containing gases that contribute to atmospheric N deposition e.g. nitrogen dioxide) and rates of N deposition in and around release pens and compare to sites lacking release pens across the seasons. We recommend this is carried out by characterising air quality emission profiles through locating measurements along transects away from release pens, to protected sites, including with measurements upwind of release pen sites, together with 'control' transects. Atmospheric concentrations need to be compared to expectations based on per bird emissions to help refine the non-agricultural emissions estimates.
- Quantify how air quality profiles change depending on release numbers, gamebird release pen locations, gamebird species, and other gamebird management-related variables. Understand whether there are context dependencies in such responses given the evidence (e.g. Madden and Sage 2020) for spatio-temporal variation in ecological effects around gamebird release sites. Further, understand whether there are non-linearities and/or thresholds in air quality responses depending on gamebird releasing variables.
- Where possible, conduct controlled experiments to disentangle the role of air quality changes due to gamebird release from other direct and associated effects that may affect ecological responses on protected sites.
- Characterise the broader air quality implications of landscape management for gamebird release, including accounting for non-linearities in behaviour depending on the scale of release, and conduct counterfactual / scenario-based analyses to assess implications of different air quality strategy recommendations.



- Understand the socio-ecological effects of gamebird releasing activities, including stakeholder perceptions, to understand the trade-offs and synergies associated with particular air quality strategy recommendations, to (i) provide an informed understanding of likely target variable trajectories of protected sites and (ii) to understand the considerations that need accounting for when developing mitigating solutions.

Given the reviewed evidence of nitrogen-related impacts from gamebird releases, it is recommended that both Natural England and the Exmoor National Park Authority adopt a more precautionary approach to determining release locations and allowable release densities, particularly in proximity to sensitive protected sites.

3.5 Profile of farming practices

The following section outlines a profile of land management within Exmoor National Park. This profile was informed by a workshop with land managers and questionnaire at an EHFN meeting. It is important to note that the land managers who provided the information for the following profile were primarily from upland farms. As a result, the findings are likely to be more representative of hill farming systems within the national park.

Livestock

Livestock farming in the national park primarily involves extensive farming of sheep and cattle, with some beef production (Table 3). Most respondents described their grazing regimes as rotational and/or extensive. The terrain and ground conditions in upland areas cannot support the same livestock numbers as other regions. The upland landscape cannot support high-intensity farming, leading to low levels of grazing on moorland. The areas west of the National Park were reported to have more dairy farms, with some possibly extending into the National Park, though there were thought to be minimal dairy farms within the park itself.

Much of the livestock raised on Exmoor is finished outside the national park.

Table 3: Average cow and sheep herd sizes from questionnaire feedback (n = 12)

Herd/flock size	Cows	Sheep
Mean	55.5	325
Min	20	0
Max	150	550

Farmers reported to be facing challenges with increasingly wetter winters, which is impacting housing and grazing practices. There is a general trend towards lower

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livestock numbers, particularly as input costs rise. The shift away from sheep farming was said to be market-driven, with some farmers reducing or dropping sheep numbers without necessarily increasing cattle numbers. Tight margins on sheep farming were leading some to maintain or increase suckler herds, with younger farmers reported to be achieving better prices for their cattle and taking risks with their herds.

It was felt that housing and grazing facilities are often outdated, not compliant with animal welfare standards, and in need of improvements for better ventilation and overall efficiency. These changes are driven by the need to meet animal welfare standards rather than increasing productivity.

Fertiliser

The application of synthetic fertilisers within the national park boundary is thought to be minimal, with a trend towards reducing usage. 64% of questionnaire respondents (n=11) reported a reduction in their usage of artificial fertiliser over the last 5/10 years. The application of nitrogen per hectare (kg/N/ha) was felt to be very low due to the cost and terrain, resulting in minimal use of NPK fertilisers. The most commonly used fertiliser is farmyard manure (FYM) (Figure 14), which is generally locally sourced. Importing or exporting manure for additional nutrients was considered impractical due to distance and fuel costs. Commonly, FYM is reserved for more productive fields where growth is better, reinforcing the extensive nature of the management practices.

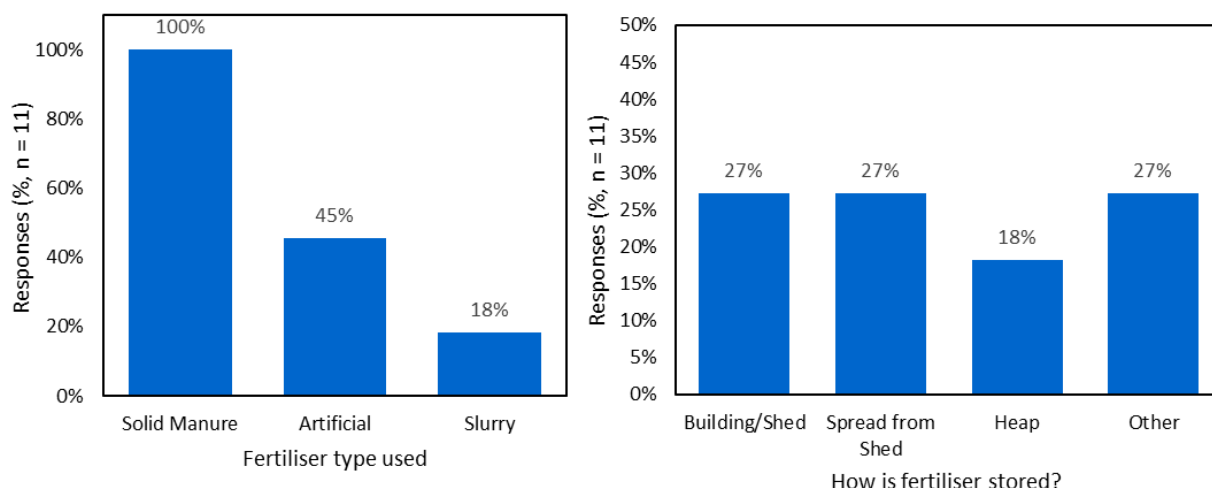


Figure 14: Questionnaire responses on type of fertiliser used and its storage.

Inside of Exmoor, beef and sheep farming is more common than dairy farming with more straw from bedding being incorporated into manure and less slurry, except on the few dairy farms within the park boundary. More slurry systems and dairy farms are found outside the national park, particularly towards the more intensive dairy farming to the west and areas within Somerset where there is more arable farming. There is generally little slurry in upland areas, and financial support for coverage is desired. Solid manure is typically spread on fields. There are a few organic farms,



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mostly in upland areas, but many farms are virtually organic without being officially registered due to the costs involved.

Soil sampling and testing are becoming more common, with younger farmers focusing on cost-effectiveness and targeted application, while the older generation tends to apply fertiliser more broadly.

Manure Storage

Manure management practices typically involve storing manure in heaps or applying it directly from sheds. Farmers face challenges in keeping rain out and preventing leaching into the ground, some farmers are planning to install covers for manure stores. All questionnaire respondents were interested in covering manure stores ($n = 10$) but stated that the main inhibitors to covering manure stores being cost and planning (Figure 15). Current best practices within the national park involve securing funding and implementing effective systems. Manure storage is a priority for compliance with farming rules for water and to improve farm efficiency. However, it is felt that atmospheric nitrogen can complicate compliance with other regulations, such as those for water.

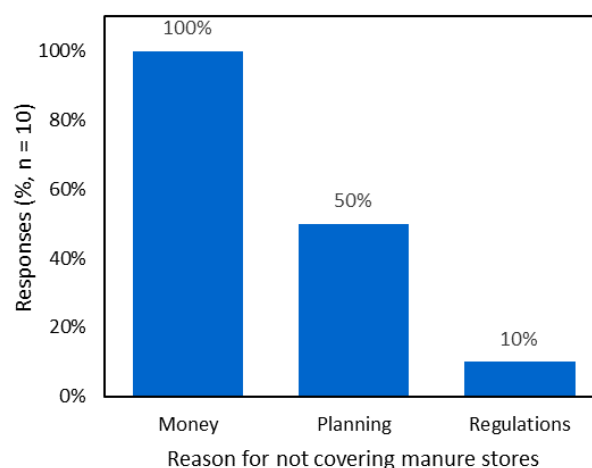


Figure 15: Questionnaire responses to question “If manure stores aren’t covered, would you like them to be and what is preventing this from being done?”

Farmers are starting to explore nitrogen-fixing crops like clover, and mid-tier Sustainable Farming Incentive (SFI) herbal layers are increasingly popular. It was felt that investments into agricultural infrastructure investments would be ideal, but finding funding was a challenge. The Countryside Stewardship capital grant for manure stores (RP28) was specifically mentioned, which requires approval from the catchment sensitive farming officer. However, as Exmoor is not a high priority of CSF, it was felt it was unlikely to receive endorsement and therefore stewardship options were limited.

Summary



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Agricultural practices within Exmoor National Park are generally lower-emission and more extensive systems. The larger nitrogen emission sources tend to be concentrated near the park's boundary or just outside it. However, certain areas within the National Park may benefit from spatially targeted mitigation measures to reduce nitrogen pollution impacts on protected sites. Areas for specific targeting could be locations of emission hotspots outlined in the Site Profile section of this report, any areas identified by any future air pollution monitoring within Exmoor and/or areas of ecological damage. Suggested measures are outlined in the Measures section of this report.

Generally, farmers and land managers expressed interest in the potential change in land management practices and different systems. From the questionnaire responses, the most interest was seen in improving manure management systems and increasing shed space (Figure 16).

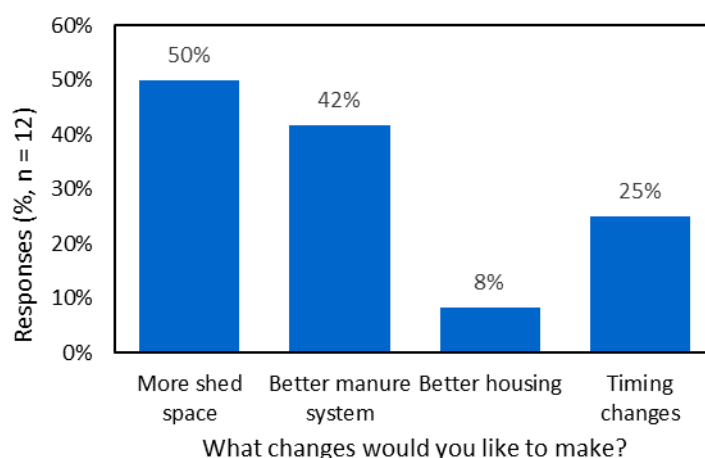


Figure 16: Questionnaire responses to question “If manure stores aren’t covered, would you like them to be and what is preventing this from being done?”

Local emission sources are the primary focus of Air Quality Management Strategies (AQMS), as nitrogen pollution disperses and reacts in the atmosphere, leading to lower concentrations further from the source. Consequently, each protected site should be assessed individually to determine its exposure to nitrogen pollution and the most effective mitigation strategies.

4. Protected Site Profiles

To understand the atmospheric N pressures for the individual designated sites, the datasets outlined in the evidence/methods section is summarised in this section for each protected site (15 SSSIs and 2 SACs) within Exmoor National Park. These data were analysed with a view to identifying key issues and suggesting potential mitigation options. The site profiles examine local spatial patterns of emission sources as well as modelled atmospheric N concentrations and deposition.

The site profiles include information on designated features and relevant exceedances of nutrient N critical loads/ NH₃ critical levels. Designated features and relevant exceedances of nutrient N critical loads/ NH₃ critical levels were summarised. An interpretation of information on local farming systems and game bird shooting/rearing in relation to their potential air quality impacts is also included where available.

Of the 15 SSSIs inside of Exmoor, five are designated for features that are either not sensitive to atmospheric N pollution or whose sensitivity is unknown. Dean Steep, Glenthorne and Hele, Samson's and Combe Martin Bays are protected as sites of geological interest, and their designated features are not sensitive to N pollution. The River Barle and River Lyn SSSIs are protected for their aquatic features. Although atmospheric N pollution can affect water quality (Rowe et al., 2024) the pathways are complex, and critical loads have not been defined for these aquatic features.

4.1 Barle Valley SSSI

The Barle Valley is located in the more agriculturally intensive southern part of Exmoor National Park, covering 652 hectares. The SSSI contains widespread tracts of ancient upland Sessile Oak woodland with variations in structure and species composition due to differences in historic management, geology and topography⁶. There are a wide range of vascular plants including many ancient woodland indicators and the lichen and bryophyte richness is of significant national and international importance. The lichen flora is exceptionally abundant and contains a number of rare species and 165 taxa of epiphytic lichens. The epiphytic lichens contain a large proportion of ancient woodland indicators giving Barle Valley a very high index of ecological continuity. Most of the SSSI is part of the Exmoor and Quantock Oakwoods SAC. Table 4 outlines the specific habitats, species, or ecosystem components that are protected or monitored for conservation within Barle Valley. For each feature, it indicated the importance of lichens and bryophytes and any relevant assigned critical level (CL_e) or critical Load (CL) for

⁶ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1000449.pdf>



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each feature. The most sensitive critical level for this SSSI is 1 µg/m³, critical loads range between 5 and 25 kg N/ha/yr (Table 4).

Table 4: Critical Loads and levels for features of Barle Valley SSSI.

Feature Name	Are Bryophytes integral?	Are Lichens Integral?	NH ₃ CL _e (µg/m ³)	Min CL Kg/N/ha/yr	Max CL Kg/N/ha/yr
Alnus Glutinosa - Fraxinus Excelsior - Lysimachia Nemorum Woodland	Yes	Yes	1 or 3		
Calluna Vulgaris - Vaccinium Myrtillus Heath	Yes	Yes	1	5	15
Combinations Of Species - Bryophytes	Yes	No	1		
Combinations Of Species - Lichens	No	Yes	1		
Cynosurus Cristatus - Centaurea Nigra Grassland	No	No	3	10	20
Cynosurus Cristatus - Centaurea Nigra Grassland	No	No	3	10	20
Quercus Spp.-Betula Spp.-Deschampsia Flexuosa Woodland	Yes	Yes	1 or 3	10	15
Quercus Robur - Pteridium Aquilinum - Rubus Fruticosus Woodland	Yes	Yes	1 or 3	15	20
Quercus Petraea - Betula Pubescens - Oxalis Acetosella Woodland	Yes	Yes	1 or 3	10	15
Quercus Petraea - Betula Pubescens - Dicranum Majus Woodland	Yes	Yes	1 or 3	10	15
Molinia Caerulea - Potentilla Erecta Mire	Yes	No	1 or 3	15	25
Mixed: Scrub, Woodland			Not sensitive		
Juncus Effusus / Acutiflorus - Galium Palustre Rush Pasture	No	No	3	15	25
Fraxinus Excelsior - Acer Campestre - Mercurialis Perennis Woodland	Yes	Yes	1 or 3	15	20
Festuca Ovina - Agrostis Capillaris - Galium Saxatile Grassland	Yes	No	1 or 3	6	10

Barle Valley is within 700m of the National Park boundary and 2km of an agricultural NH₃ emission hotspot (Figure 17). Agricultural emission density increases with distance from the site. Key emission sources within 2km of the site are mainly beef, with some sheep. Beyond 2 km, dairy farming becomes a notable



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emission source. Dairy farming is possibly also an emission source within 1 km of the site boundary, however there are too few farms present in the dataset to quantify, hence aggregated with “other sources”, to avoid showing disclosive data. There are also likely gamebird releases nearby.

Ammonia concentrations for most of Barle Valley SSSI are modelled to be between 1 and 1.5 $\mu\text{g NH}_3 \text{ m}^{-3}$ (Figure 18), with an area-weighted average NH_3 concentrations estimated at 1.2 $\mu\text{g NH}_3 \text{ m}^{-3}$ (Figure 17). This is above the critical level for lichens and bryophytes which has been assigned to the site as it is home to many national and internationally important lichen and bryophyte species. With higher ammonia concentrations and local emission sources, this site has the potential to benefit from some targeted measures. There appears to be a north-south gradient of ammonia concentrations at the site, with the northern parts seeing modelled concentrations of below 1 $\mu\text{g NH}_3 \text{ m}^{-3}$. This is in line with the agricultural emissions data showing hotspots to the south of the site and the general trend towards higher atmospheric N pollution in the areas immediately surrounding the Exmoor NP boundary.

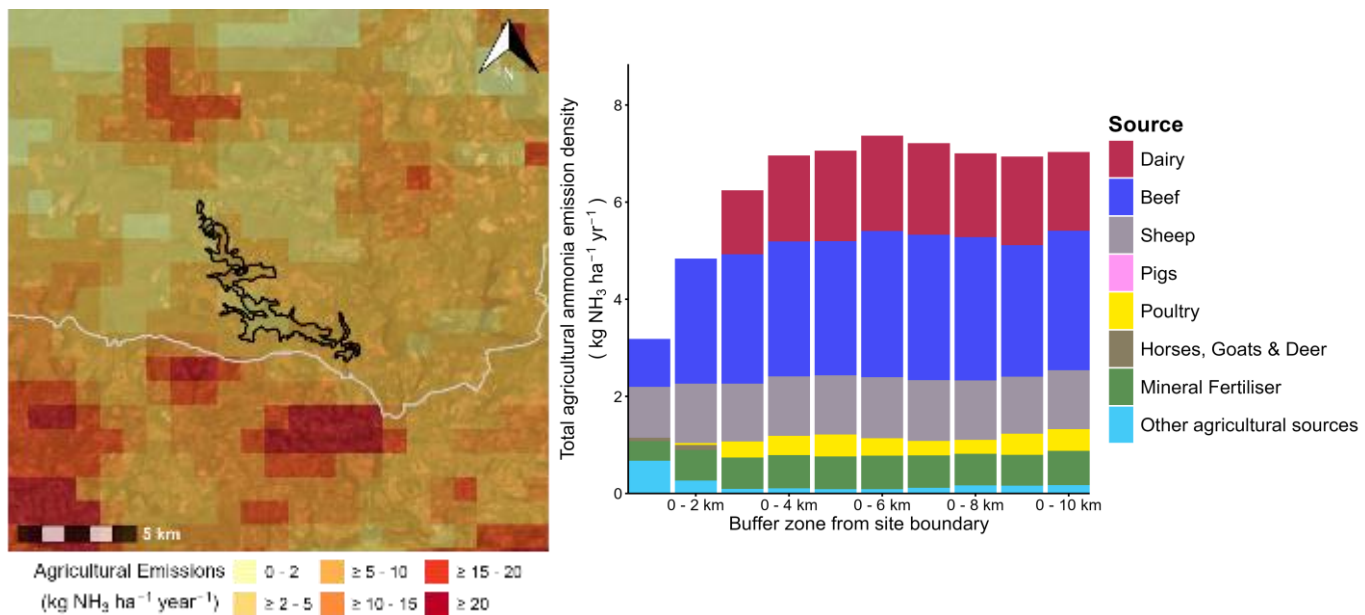


Figure 17: Gridded 2021 agricultural emission estimates (1km x 1km grid resolution, NAEI) (LEFT) and agricultural emission density estimates (RIGHT) for Barle Valley SSSI. The category ‘other agricultural sources’ refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).



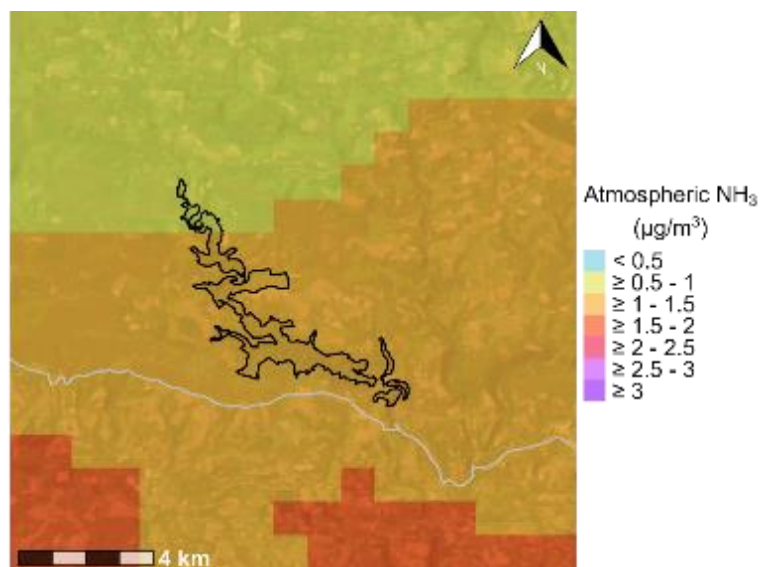


Figure 18: Mean modelled NH₃ concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Barle Valley SSSI

Only nitrogen deposition estimates onto woodland features are described in the following section, as this matches the woodland habitats in the SSSI. This also matches the Annex 1 Old sessile oak woods habitat classified by the JNCC for the Exmoor and Quantock Oakwoods SAC, which most of this site is a part of. The area-weighted mean rate of N deposition to woodland vegetation (where present) to the site is estimated at ca. 23 kg N ha⁻¹ yr⁻¹, with ~ 68 % of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 19).

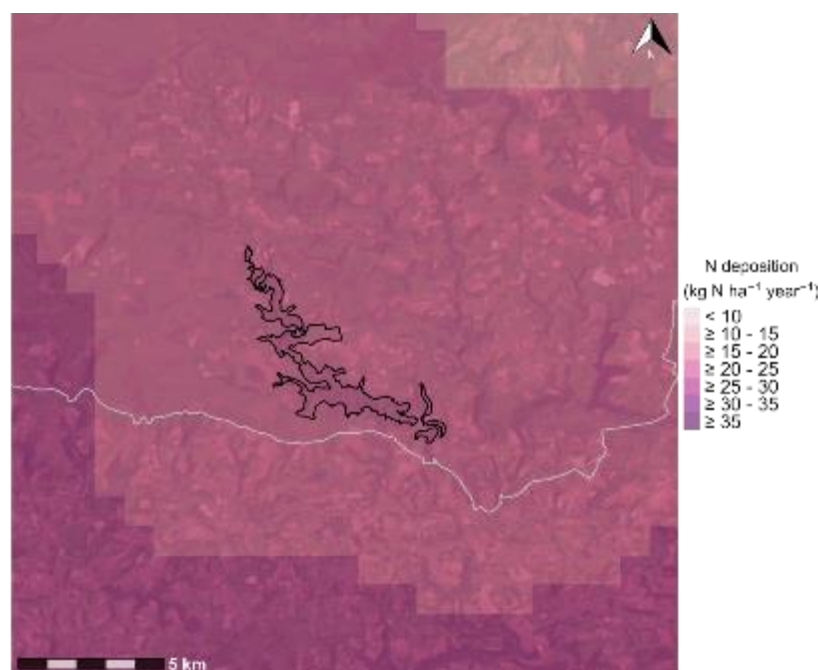


Figure 19: CBED mean modelled N deposition for 2019 – 2021 onto woodland features for Barle Valley SSSI

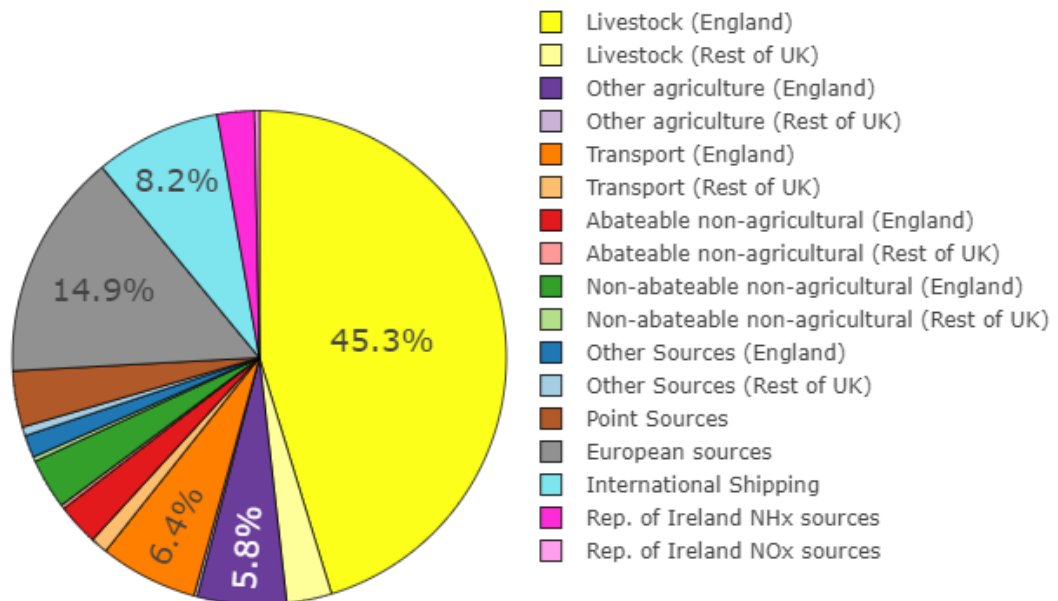


Figure 20: Source Attribution (2018) total N deposition received by woodland features for Barle Valley SSSI.

The 2018 source attribution dataset estimates that the largest contribution to nitrogen deposition (to woodland features) received by Barle Valley is from livestock emission sources in England (~45%, Figure 20). Livestock sources from across the rest of the UK contribute a relatively small proportion and given the location of the site, these livestock sources from the rest of the UK are likely to primarily originate from sources within Wales. Transport contributes ~7% of the total N deposition received.

A substantial proportion of nitrogen deposition is also estimated to originate from sources outside the UK, with European sources contributing ~15%, and international shipping ~8%. These sources would require international intervention to reduce their contribution to N deposition received by the site.

4.2 Dean Steep SSSI

Dean Steep is a very small SSSI covering 1.9ha located to the north west of Exmoor national park. It has designated site status as a site of geological interest due to the exposures of the Devonian aged 'Lynton Beds' and the overlying 'Hangman Sandstone' formations⁷. There is no assigned critical level or load to this site as there are no N-sensitive features. The site is not part of either of the SACs within Exmoor.

⁷ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1004236.pdf>

4.3 Dunster Park and Heathland SSSI

Dunster Park and Heathland SSSI is located to the north-east of Exmoor National Park, intersecting with the park boundary. The site is notified for nationally important lowland dry heath, dry lowland acid grassland, wood-pasture with veteran trees and ancient semi-natural oak woodland habitats. The fauna of the lowland heath includes a nationally rare butterfly, the heath fritillary *Mellicta athalia*⁸. The site has been assigned a critical level of $1\mu\text{g NH}_3\text{ m}^{-3}$ for its heath and grassland features. The most sensitive critical level for this SSSI is $1\mu\text{g/m}^3$, critical loads range between 5 and 20 kg N/ha/yr (Table 5).

Table 5: Critical Loads and levels for features of Dunster Park and Heathland SSSI.

Feature Name	Any Bryophytes integral?	Are Lichens Integral?	NH ₃ CL _e ($\mu\text{g/m}^3$)	Minimum CL (KgN/ha/yr)	Maximum CL (KgN/ha/yr)
Festuca Ovina - Agrostis Capillaris - Galium Saxatile Grassland	Yes	No	1 or 3	6	10
Festuca Ovina - Agrostis Capillaris - Rumex Acetosella Grassland	Yes	Yes	1	5	15
Quercus Robur - Pteridium Aquilinum - Rubus Fruticosus Woodland	Yes	Yes	1 or 3	15	20
Quercus Spp.-Betula Spp.-Deschampsia Flexuosa Woodland	Yes	Yes	1 or 3	10	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15

In line with feedback from land managers, the latest NAEI agricultural emission maps (2021) show more intensive emission sources in this part of the national park and in close proximity to the site (Figure 21), there are fewer than 5 farms for some emission source categories within 2km of the site boundary, meaning that these had to be aggregated into 'other agricultural sources' to meet the confidentiality threshold. Notable pig and dairy emission sources appear within the 0 - 3km buffer zone from the site boundary. Pig farms tend to be high intensity emissions which may impact the SSSI given its close proximity to the national park boundary.

There appears to be a small east to west gradient in ammonia concentrations at Dunster Park and Heathland SSSI, as the west of the site is modelled to have a lower concentration of below $1\mu\text{g NH}_3\text{ m}^{-3}$ whereas the centre and east are between 1 and $1.5\mu\text{g NH}_3\text{ m}^{-3}$ (Figure 22). The SSSI has an estimated area-weighted average NH₃ concentration of $1.1\mu\text{g NH}_3\text{ m}^{-3}$, just above the critical level

⁸ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/2000415.pdf>

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for lichens and bryophytes. With higher ammonia concentrations and local emission sources, this site has potential to benefit from some targeted measures.

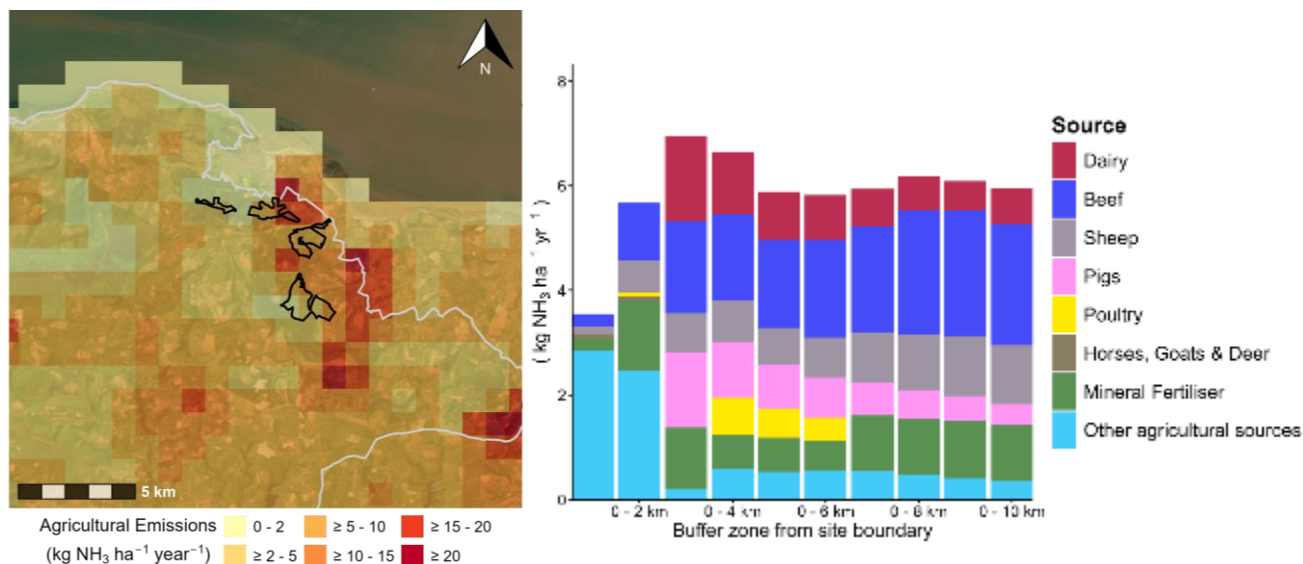


Figure 21: Gridded 2021 agricultural emission estimates (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for Dunster Park and Heathland SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

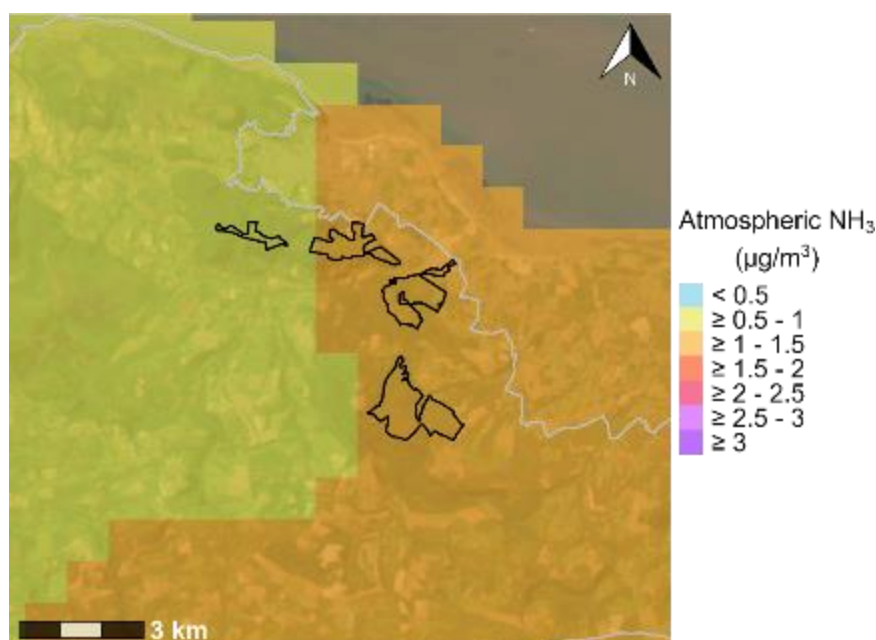


Figure 22: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Dunster Park and Heathland SSSI

Only N deposition estimates onto low-growing semi-natural vegetation are described in the following section, since this is appropriate for the heath and



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grassland habitats in the SSSI. The area-weighted mean rate of N deposition to open vegetation (where present) to the site is relatively low at $\sim 12 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, with $\sim 60\%$ of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021 (Figure 23).

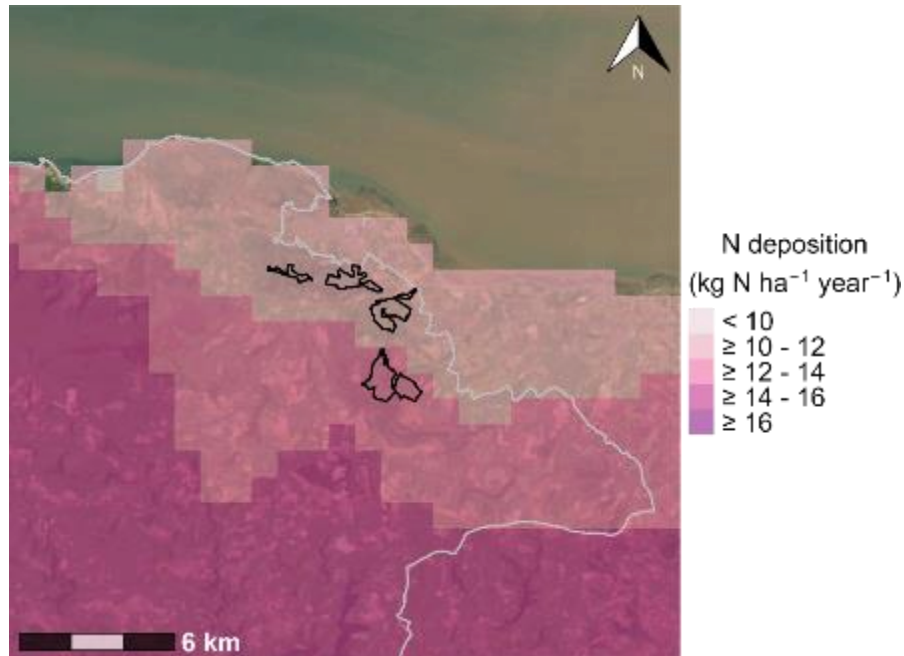


Figure 23: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for Dunster Park and Heathland SSSI

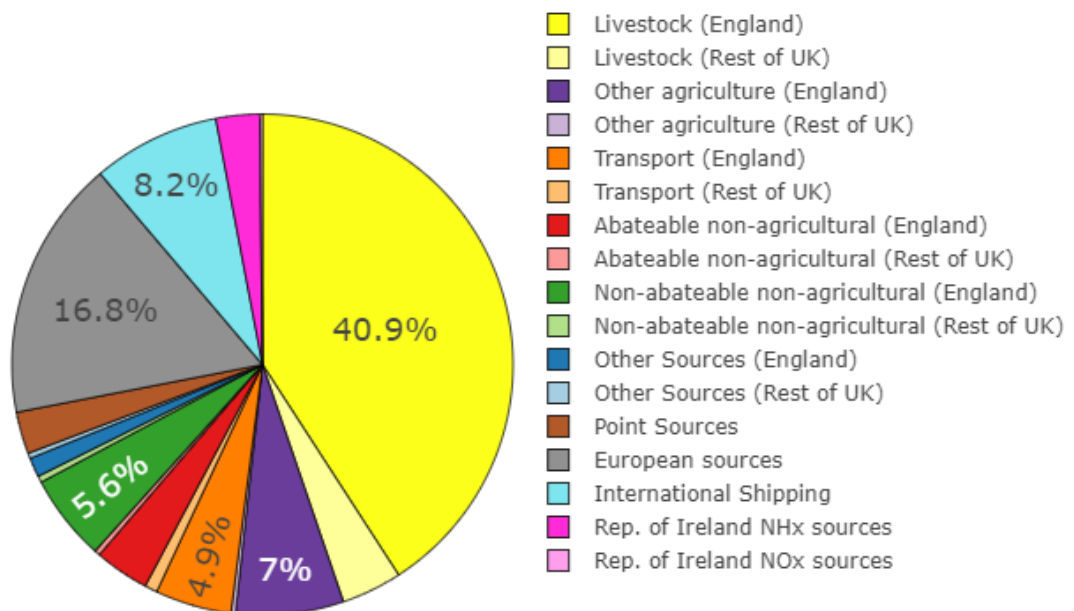


Figure 24: Source Attribution (2018) Total N deposition received by low-growing semi-natural vegetation for Dunster Park

The 2018 source attribution dataset estimates that the largest source of N deposition (onto open vegetation) received by Dunster Park and Heathland SSSI is

livestock emission sources in England (~41%, Figure 24). Other agriculture from England (i.e. emissions from mineral fertiliser application) contributed a further 7% ,and transport contributed ~6% of the total N deposition received.

A substantial proportion of nitrogen deposition is estimated to originate from sources outside the UK, with European sources contributing the second largest proportion (~17%), followed by international shipping (~8%). These sources would require international interventions to reduce their contribution to N deposition received by the site.

4.4 Exmoor Coastal Heaths SSSI

Exmoor Coastal Heaths SSSI is an extensive area (1,732 ha) of rare heathland communities located along the coastline of Exmoor National Park. The sites are important for the presence, range and transitions between habitats including upland heath, mires and grassland. At lower altitudes and in the coastal zone further important habitats occur including acidic and maritime grassland⁹. Associated particularly with the coastal communities and woods are a wide range of nationally rare and scarce plants. Exmoor Coastal Heath SSSI heavily overlaps with the coastal areas of the Exmoor Heaths SAC.

Table 6: Critical Loads and levels for features of Exmoor Coastal Heaths SSSI

Feature Name	Any Bryophytes Integral?	Are Lichens Integral?	NH ₃ CL _e (µg/m ³)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
Calluna Vulgaris - Erica Cinerea Heath	Yes	Yes	1	5	15
Calluna Vulgaris - Ulex Gallii Heath	Yes	No	1 or 3	5	15
Agrostis Curtisii Grassland	Yes	No	1 or 3	6	10
Alnus Glutinosa - Fraxinus Excelsior - Lysimachia Nemorum Woodland	Yes	Yes	1 or 3		
Calluna Vulgaris - Vaccinium Myrtillus Heath	Yes	Yes	1	5	15
Carex Dioica - Pinguicula Vulgaris Mire	Yes	No	1	15	25
Festuca Ovina - Agrostis Capillaris - Galium Saxatile Grassland	Yes	No	1 or 3	6	10
Festuca Ovina - Agrostis Capillaris - Rumex Acetosella Grassland	Yes	Yes	1	5	15
Festuca Ovina - Agrostis Capillaris - Rumex Acetosella Lowland Acid Grassland	Yes	Yes	1	6	10

⁹ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1004152.pdf>

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Festuca Rubra - Armeria Maritima Maritime Grassland	No	No	1 or 3		
Festuca Rubra - Hyacinthoides Non-Scripta Maritime Bluebell Community	No	No	1 or 3		
Hypericum Elodes - Potamogeton Polygonifolius Soakway	Yes	No	1	5	15
Juncus Effusus / Acutiflorus - Galium Palustre Rush Pasture	No	No	3	15	25
Molinia Caerulea - Potentilla Erecta Mire	Yes	No	1 or 3	15	25
Quercus Petraea - Betula Pubescens - Dicranum Majus Woodland	Yes	Yes	1 or 3	10	15
Quercus Petraea - Betula Pubescens - Oxalis Acetosella Woodland	Yes	Yes	1 or 3	10	15
Quercus Spp.-Betula Spp.-Deschampsia Flexuosa Woodland	Yes	Yes	1 or 3	10	15
Scirpus Cespitosus - Erica Tetralix Wet Heath	Yes	Yes	1	5	15
U4/20-Related Species Rich Bracken	No	No	3	6	10
Ulex Europaeus - Rubus Fruticosus Scrub	No	No	3	10	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Vascular Plant Assemblage			3		

In terms of agricultural NH₃ emissions, each of the three dispersed sub-areas of Exmoor Coastal Heaths SSSI have different emission sources nearby. The westernmost part is in an area of higher emissions reaching over 20 kg NH₃ ha⁻¹ yr⁻¹ within a few kilometres of the national park boundary (Figure 25). The central and easternmost parts are estimated to be in areas of lower agricultural emission density. Agricultural emission density increases with distance from the SSSI (Figure 26). The main emission sources within 2 km of the site are beef and sheep, with a growing impact of dairy with increasing distance from the site and contributions from pig farming.

Overall, the SSSI has an estimated area-weighted average NH₃ concentration of 0.9 µg NH₃ m⁻³ yr⁻¹, below the critical level for the most sensitive features on the site (1 µg). The spatial patterns of NH₃ concentrations follow a similar pattern to agricultural ammonia emissions, with the westernmost area of the SSSI estimated to exceed 1 µg NH₃ m⁻³ yr⁻¹ (Figure 27).



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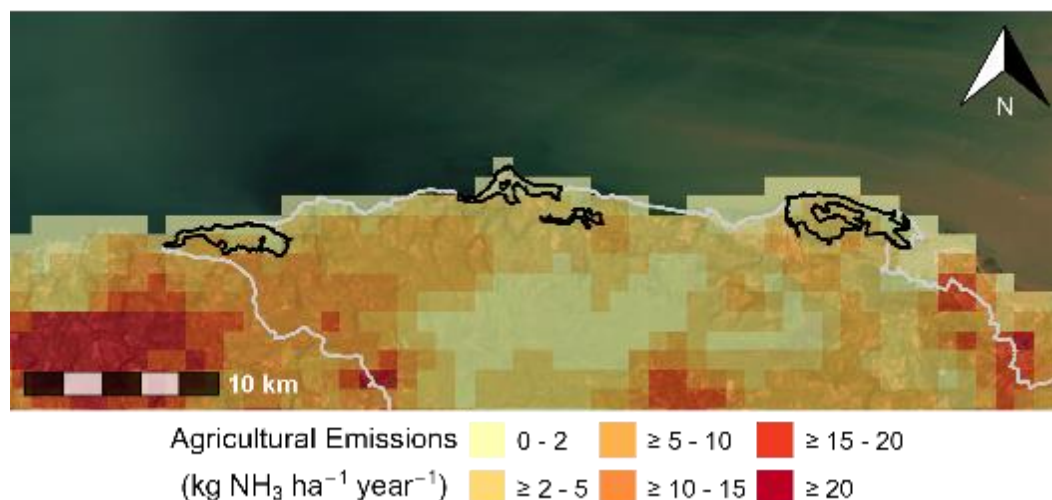


Figure 25: Gridded 2021 agricultural emission estimates for Exmoor Coastal Heaths SSSI (1km x 1km grid resolution, NAEI)

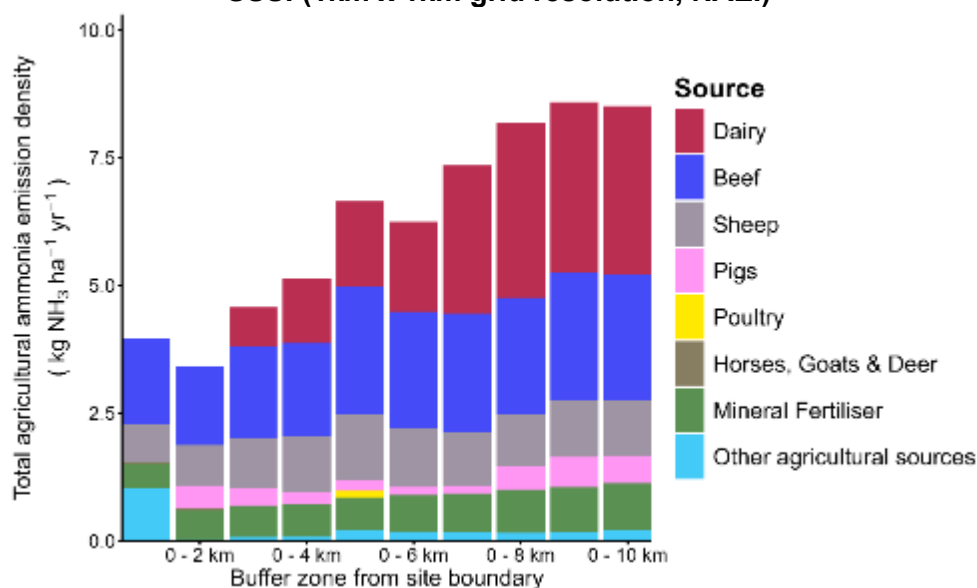


Figure 26: Agricultural emission density estimates for Exmoor Coastal Heaths SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).



Figure 27: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Exmoor Coastal Heaths SSSI



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Only nitrogen deposition estimates onto open semi-natural vegetation will be described in the following section, as this is appropriate for the heath and grassland habitats in the SSSI. The area-weighted mean rate of N deposition onto open vegetation (where present) to the site is relatively low at $11.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, with ~53% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 28).

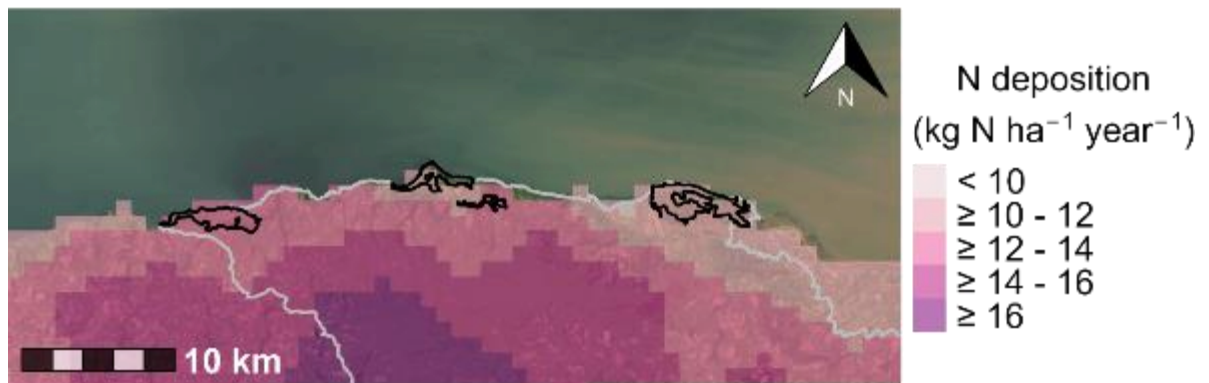


Figure 28: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for Exmoor Coastal Heaths SSSI

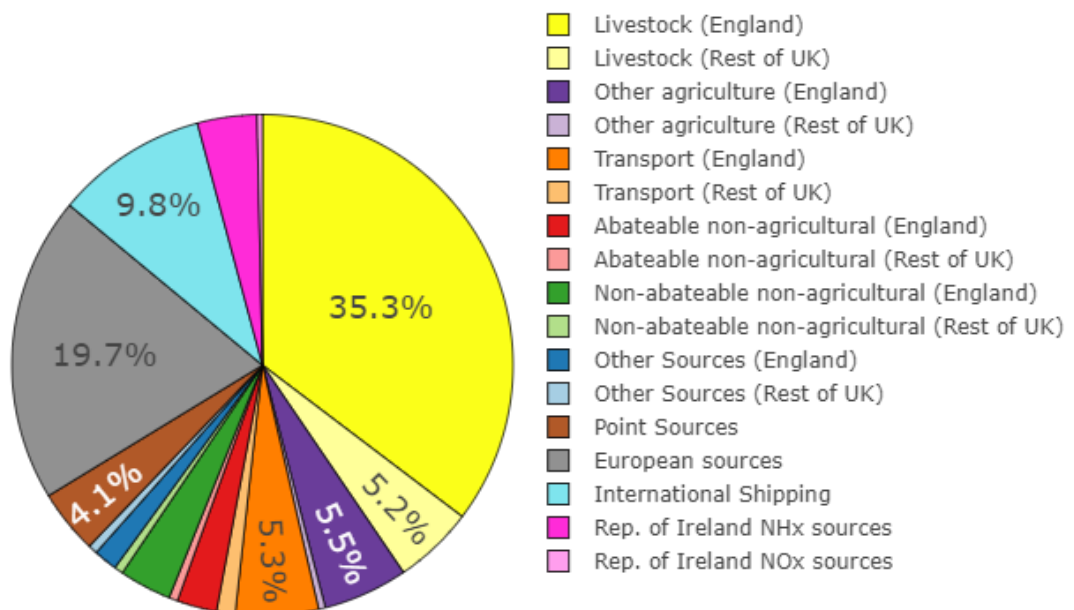


Figure 29: Source Attribution (2018) Total N deposition received by low-growing semi-natural vegetation for Exmoor Coastal Heaths

This site has a much more varied range of sources of N deposition than others within the national park (Figure 29). The 2018 source attribution dataset estimates that the largest source of N deposition (to low-growing vegetation) received by Exmoor Coastal Heaths SSSI is livestock in England (~35%). Together, international shipping and European emission sources contribute almost 30% of the total N deposition.



4.5 Glenthorne SSSI

Glenthorne is a small site (13 ha) located in the centre of Exmoor's coastline. It is designated for features of geological interest for its access to a unique geological formation (Hangman Sandstone Group)¹⁰. There is no assigned critical level or load to this site as there are no N-sensitive features. The site is not part of either of the SACs within Exmoor.

4.6 Hele, Samson's and Combe Martin Bays SSSI

Hele, Samson's and Combe Martin Bay is an SSSI designated for features of geological interest, and has no N-sensitive features¹¹.

4.7 Nettlecombe Park SSSI

Nettlecombe Park SSSI has been wood pasture or parkland for at least 400 years, with some trees over 200 years old. Nationally rare and important epiphytic lichens have established and continue to live on the trees at this SSSI¹². The site spans 108 ha and lies near the eastern edge of the National Park. Nettlecombe Park SSSI does not overlap with either of the two SACs. One feature of Nettlecombe Park has a critical level of 1 $\mu\text{g}/\text{m}^3$ assigned, there are no critical loads.

According to the agricultural NH_3 emissions map from the NAEI (Figure 30), there are relatively small emissions in closer vicinity to the Nettlecombe Park, with a higher emission zone to the SE of the site that appears to be located largely but not entirely outside of the NP boundary. This is reflected in the agricultural emission density graphs, which shows a relatively low density for the first 3 kilometres around Nettlecombe Park SSSI (Figure 30), on average, despite the slightly higher emission zone (up to $>20 \text{ kg } \text{NH}_3 \text{ ha}^{-1} \text{ yr}^{-1}$) in the SE quadrants of the 2 and 3 km zones of the map. Within 2 km of the SSSI boundary, mineral fertiliser is the greatest agricultural emission source. Beef and dairy have a growing influence on emission density with increasing distances from the site. Larger emission hotspots of over $20 \text{ kg } \text{NH}_3 \text{ ha}^{-1} \text{ yr}^{-1}$ are modelled $>2 \text{ km}$ southeast from the site near Elworthy.

All of the SSSI is modelled to have ammonia concentrations between 1 and $1.5 \mu\text{g } \text{NH}_3 \text{ m}^{-3}$ (Figure 31). The SSSI has an estimated area-weighted average NH_3 concentration of $1.2 \mu\text{g } \text{NH}_3 \text{ m}^{-3}$, just above the critical level for lichens and

¹⁰ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1002137.pdf>

¹¹ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/2000117.pdf>

¹² <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1005670.pdf>



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bryophytes which has been assigned to the site as it is home to many important lichen species.

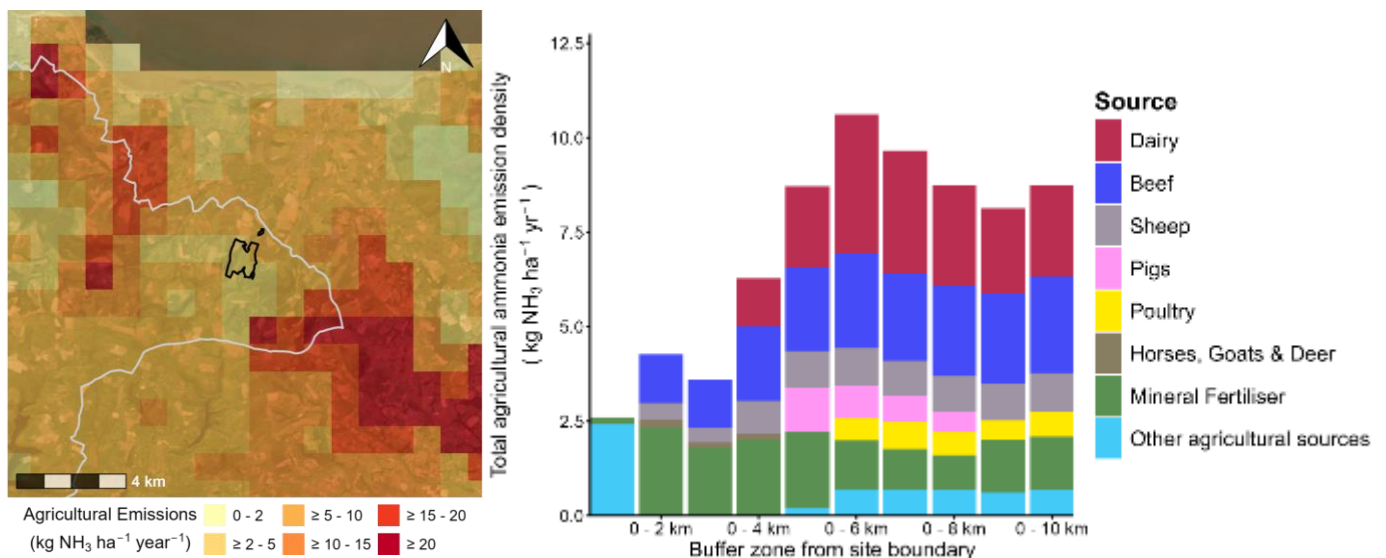


Figure 30: Gridded 2021 agricultural emission estimates (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for Nettlecombe Park SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

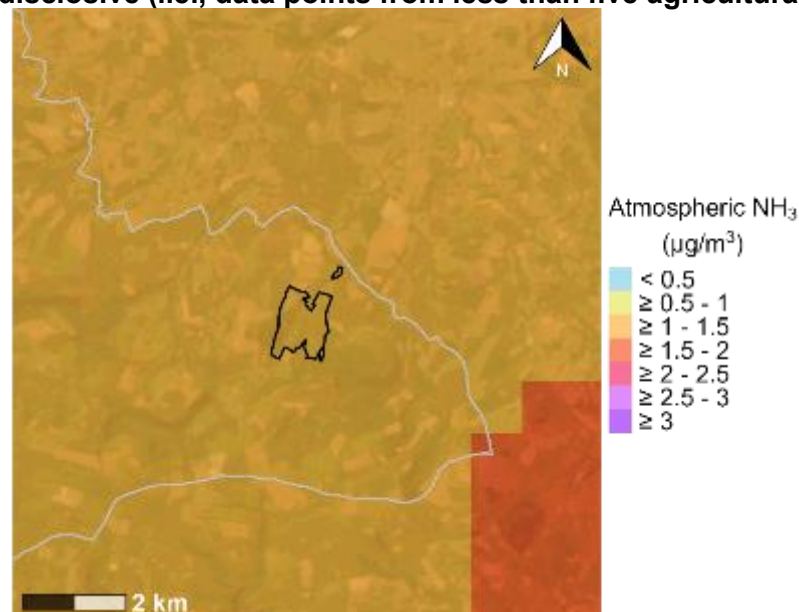


Figure 31: Mean modelled NH₃ concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Nettlecombe Park SSSI

Only N deposition estimates onto woodland features are described in the following section, as this matches the designated woodland features in the SSSI. The area-weighted mean rate of N deposition to woodland vegetation (where present) to the site is ~21 kg N ha⁻¹ yr⁻¹, with ~ 76 % of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 32). This higher

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level of dry deposition suggests greater potential for reduction in N deposition from implementation of local measures for this site.

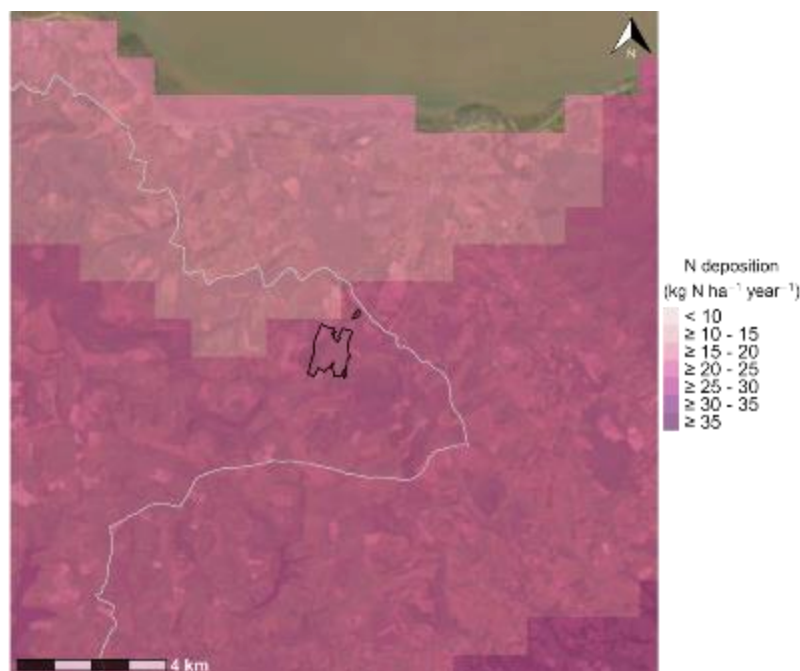


Figure 32: CBED mean modelled N deposition for 2019 – 2021 onto woodland features, for Nettlecombe Park SSSI

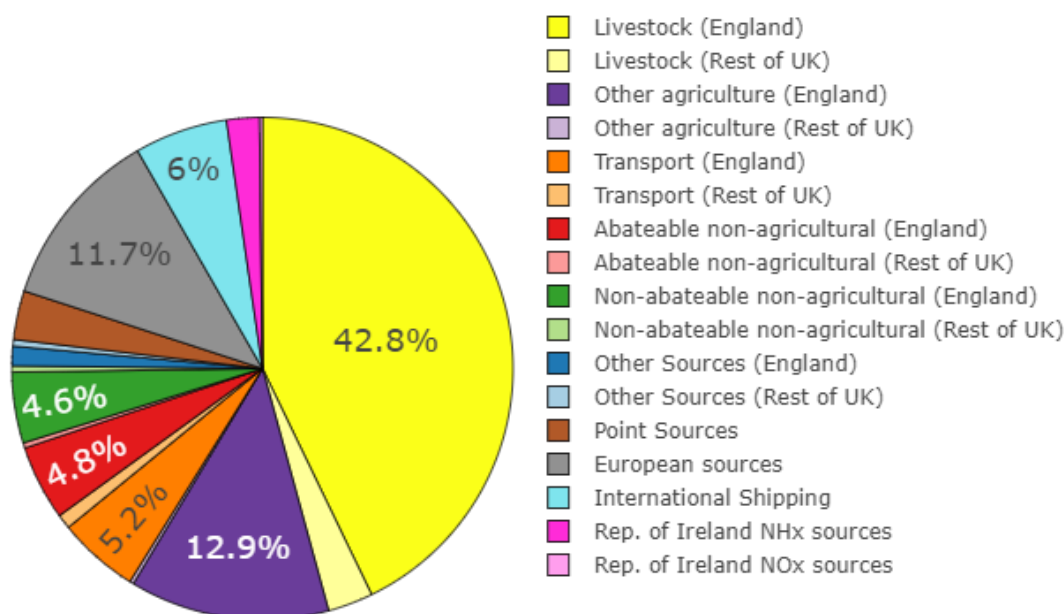


Figure 33: Source Attribution (2018) Total N deposition received by woodland features for Nettlecombe

The 2018 source attribution dataset estimates that the largest source of N deposition (to woodland features) received by Nettlecombe Park SSSI is livestock emission sources in England (~43%, Figure 33). Transport sources in England contribute ~5% of the total N deposition received. This site has a higher proportion

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of N deposition from other agriculture, i.e. from mineral fertiliser application in England (13%).

A notable proportion of N deposition is estimated to originate from sources outside the UK, with European sources contributing ~12% and international shipping ~6%. These sources would require international intervention to reduce their contribution to N deposition received by the site.

4.8 North Exmoor SSSI

North Exmoor is a very large SSSI (12,000 hectares) in the centre of Exmoor National Park. The site is nationally important for its lowland heath communities, which are rare in this region of Britain¹³. The site is also important for its transitions from blanket mires to upland heaths and to ancient semi-natural woodlands which are home to exceptional woodland lichen flora. The site is split into two main blocks: the major one to the North and a smaller one by Simonsbath to the south/southwest. A large area of the northern block is part of the Exmoor Heaths SAC.

Table 7 Critical Loads and levels for features of North Exmoor SSSI

Feature Name	Are Bryophytes integral?	Are Lichens Integral?	NH ₃ CLe (µg/m ³)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
Quercus Robur - Pteridium Aquilinum - Rubus Fruticosus Woodland	Yes	Yes	1 or 3	15	20
Quercus Petraea - Betula Pubescens - Oxalis Acetosella Woodland	Yes	Yes	1 or 3	10	15
Ranunculus Omiophyllus - Montia Fontana Rill	Yes	No	1	5	15
Scirpus Cespitosus - Erica Tetralix Wet Heath	Yes	Yes	1	5	15
Scirpus Cespitosus - Eriophorum Vaginatum Blanket Mire	Yes	Yes	1	5	10
Sphagnum Cuspidatum/recurvum (Fallax) Bog Pool Community	Yes	No	1	5	10
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Alnus Glutinosa - Fraxinus Excelsior - Lysimachia Nemorum Woodland	Yes	Yes	1 or 3		
Calluna Vulgaris - Vaccinium Myrtillus Heath	Yes	Yes	1	5	15
Carex Dioica - Pinguicula Vulgaris Mire	Yes	No	1	15	25

¹³ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1006541.pdf>



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Carex Echinata - Sphagnum Recurvum (Fallax) /Auriculatum (Denticulatum) Mire	Yes	No	1	5	15
Combinations Of Species - Lichens	No	Yes	1		
Erica Tetralix - Sphagnum Compactum Wet Heath	Yes	Yes	1	5	15
Festuca Ovina - Agrostis Capillaris - Galium Saxatile Grassland	Yes	No	1 or 3	6	10
Lowland Heath			Not sensitive		
Quercus Spp.-Betula Spp.-Deschampsia Flexuosa Woodland	Yes	Yes	1 or 3	10	15
Quercus Petraea - Betula Pubescens - Dicranum Majus Woodland	Yes	Yes	1 or 3	10	15
Narthecium Ossifragum - Sphagnum Papillosum Mire	Yes	No	1	5	15
Molinia Caerulea - Potentilla Erecta Mire	Yes	No	1 or 3	15	25
Hypericum Elodes - Potamogeton Polygonifolius Soakway	Yes	No	1	5	15
Juncus Effusus / Acutiflorus - Galium Palustre Rush Pasture	No	No	3	15	25

In terms of agricultural NH₃ emissions, the NAEI data suggest that North Exmoor SSSI has low levels of agricultural emissions inside the designated site. This is in line with feedback from hill farm managers who report mainly extensive and low intensity grazing practices on upland heaths. There are a two areas of higher agricultural ammonia emissions near the SSSI boundary – one within the National Park near Exford to the south of the heathlands and one to the southwest of the site near Barton Town.

Agricultural emission density generally increases with distance from the site (Figure 34). There are no clear dominant emission sources within the first few concentric buffer zones outside of the site. The main emission sources tend to be beef and sheep which are representative of extensive upland farming. There is some potential pig farming within 1km of site boundary but this is not a dominant emission source.



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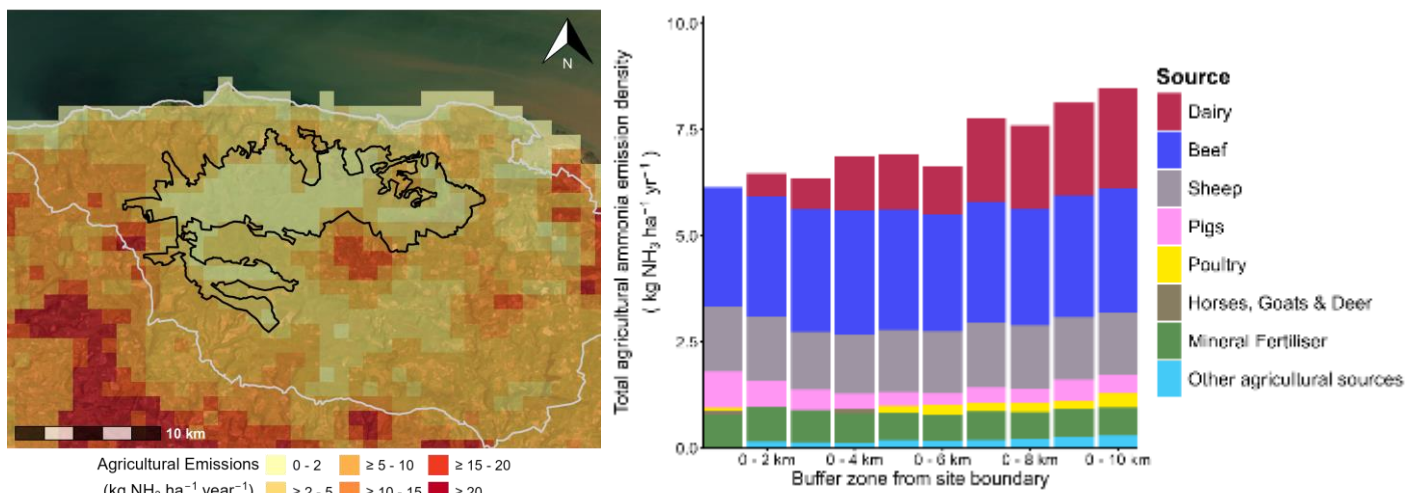


Figure 34: Gridded 2021 agricultural emission estimates SSSI (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for North Exmoor SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

Most of the North Exmoor SSSI are modelled to experience concentrations below $1 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$, the critical level for lichens and bryophyte species present on the site (**Figure 35**). There are some areas in the southwestern zone which are on the border of being above this threshold. Overall, the estimated area-weighted average NH_3 concentration is $0.8 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$.

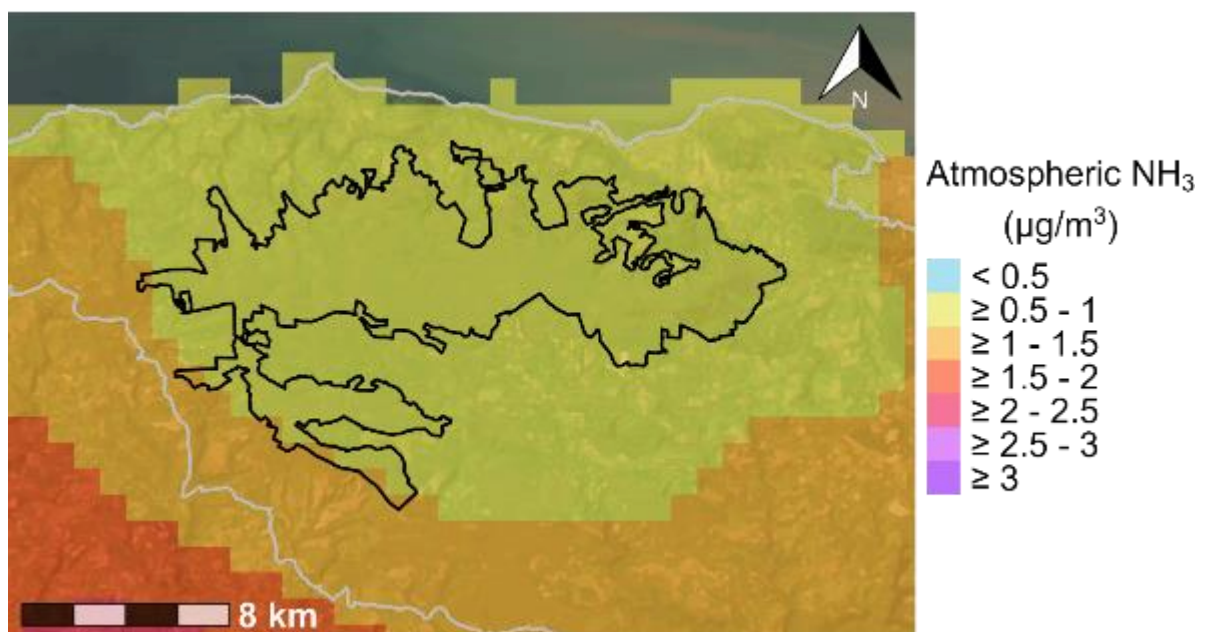


Figure 35: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for North Exmoor SSSI

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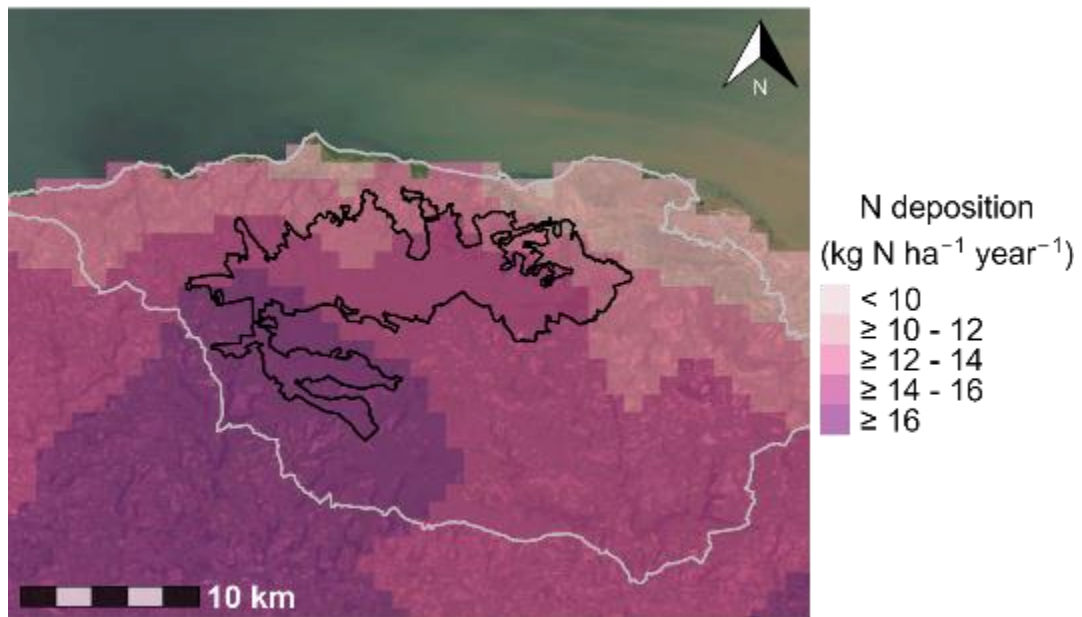


Figure 36: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for North Exmoor

As the site contains both ecologically important woodland features and low-growing vegetation features in the heathlands, both deposition types will be considered.

The area-weighted mean rate of N deposition to low-growing semi-natural vegetation (where present) to the site is 15 kg N ha⁻¹ yr⁻¹, with ~ 45% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, **Figure 36**).

North Exmoor SSSI has a varied range of source sectors contributing to N deposition (Figure 37). The 2018 source attribution dataset estimates that the largest source of N deposition (to low-lying vegetation) received by North Exmoor SSSI is from livestock emissions in England (~37%). Together, international shipping and European emission sources contribute to over 30% of total N deposition.

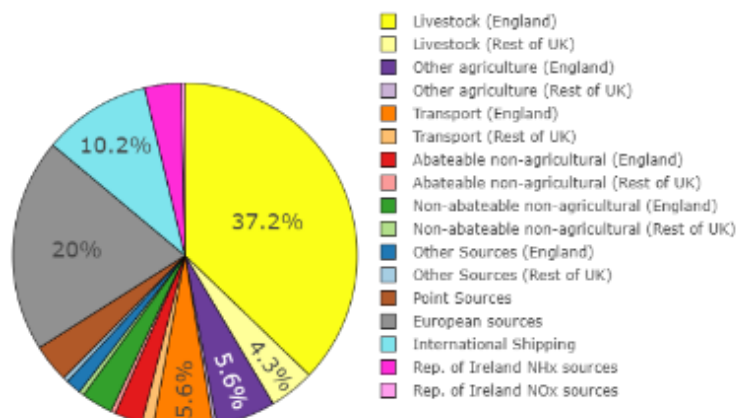


Figure 37: Source Attribution (2018) Total N deposition received by low-growing semi-natural vegetation for North Exmoor

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The area-weighted mean rate of N deposition to woodland (where present) to the site is 22 kg N ha⁻¹ yr⁻¹, with ~ 58% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 38). N deposition is higher in the southwestern zone of the SSSI.

The 2018 source attribution dataset estimates that the largest source of N deposition (to woodlands) received is livestock emission sources in England (~39%). The second largest N deposition source is transboundary pollution from European sources (~17%), followed by International shipping (~10%) and transport emission sources from within England (~7%).

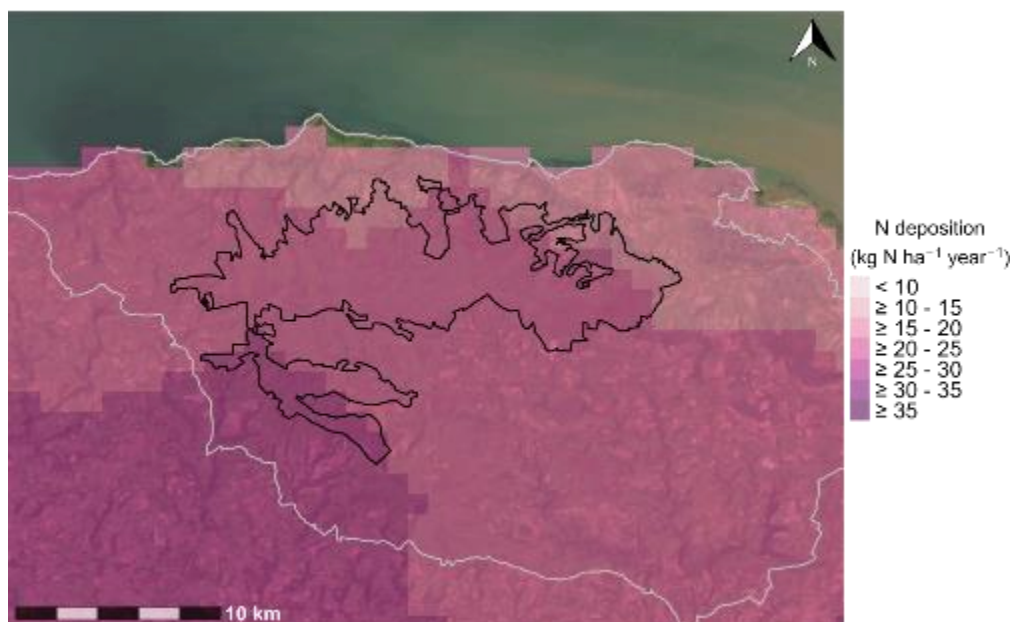


Figure 38: CBED mean modelled N deposition for 2019 – 2021 onto woodland features, for North Exmoor SSSI

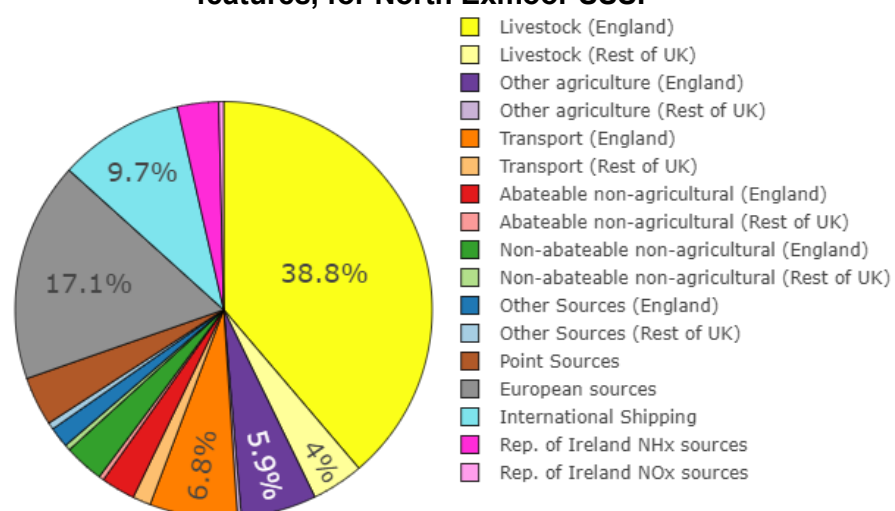


Figure 39: Source Attribution (2018) Total N deposition received by woodland features for North Exmoor SSSI

4.9 Pinkworthy and Driver Farm SSSI

Pinkworthy and Driver Farm SSSI is split into two parts, both of which support acidic grassland and wetland vegetation. The site is home to nationally important grassland fungi (waxcap species)¹⁴. The species present mean that the site has been assigned a critical level of 1 or 3 $\mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$ (APIS).

Table 8 Critical Loads and levels for features of Pinkworthy and Driver Farm SSSI

Feature Name	Any Bryophytes integral?	Are Lichens Integral?	NH ₃ CLe ($\mu\text{g}/\text{m}^3$)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
Festuca Ovina - Agrostis Capillaris - Galium Saxatile Grassland	Yes	No	1 or 3	6	10
Juncus Effusus / Acutiflorus - Galium Palustre Rush Pasture	No	No	3	15	25
Grassland fungi			1 or 3		

The emission maps from the NAEI suggest low levels of agricultural emissions within the site, below 5 kg $\text{NH}_3 \text{ ha}^{-1} \text{ yr}^{-1}$ (Figure 40). The closest area of higher emission is ~2 km to the west of the site near the National Park Boundary. Agricultural emission density generally increases with distance from the site (Figure 40), although density remains relatively low on average. Agricultural emission sources tend to be beef and sheep farming which is more representative of the extensive upland farming prevalent within the National Park. There are some potential pig emissions within 2 km of the site boundary, however these are relatively small and are not considered a dominant emission source.

¹⁴ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/2000867.pdf>

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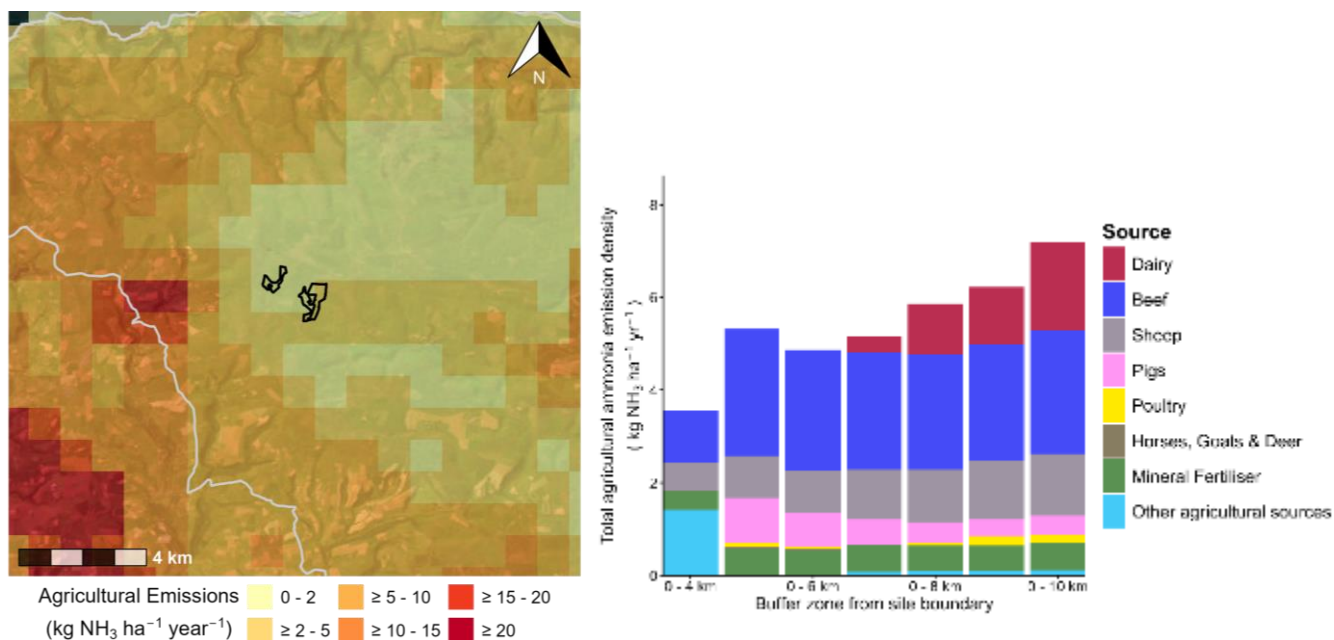


Figure 40: Gridded 2021 agricultural emission estimates (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for Pinkworthy and Driver Farm SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

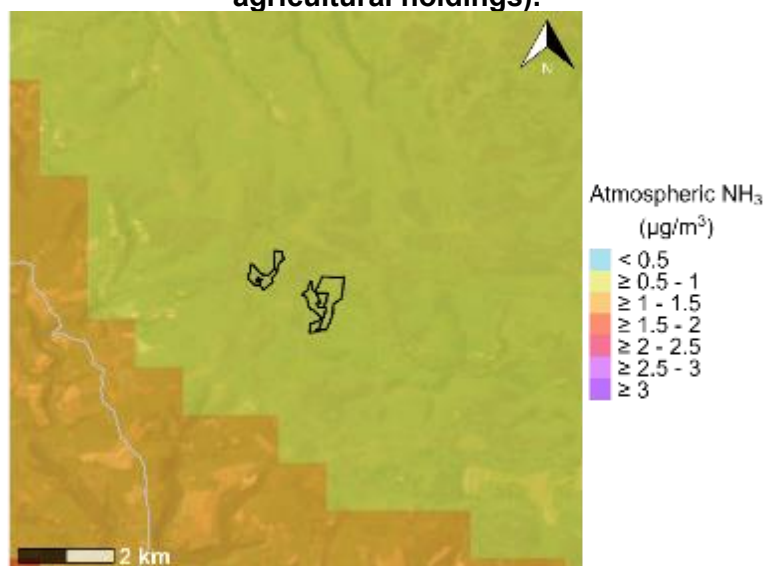


Figure 41: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Pinkworthy and Driver Farm SSSI

All of the SSSI is modelled to have ammonia concentrations below $1 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$, below the critical level assigned to the site's features (Figure 41). Overall, the estimated area-weighted average NH_3 concentration is $0.8 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$.

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The area-weighted mean rate of N deposition to low-growing semi-natural vegetation (where present) on the site is @17 kg N ha⁻¹ yr⁻¹, with ~ 45% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, **Figure 42**).

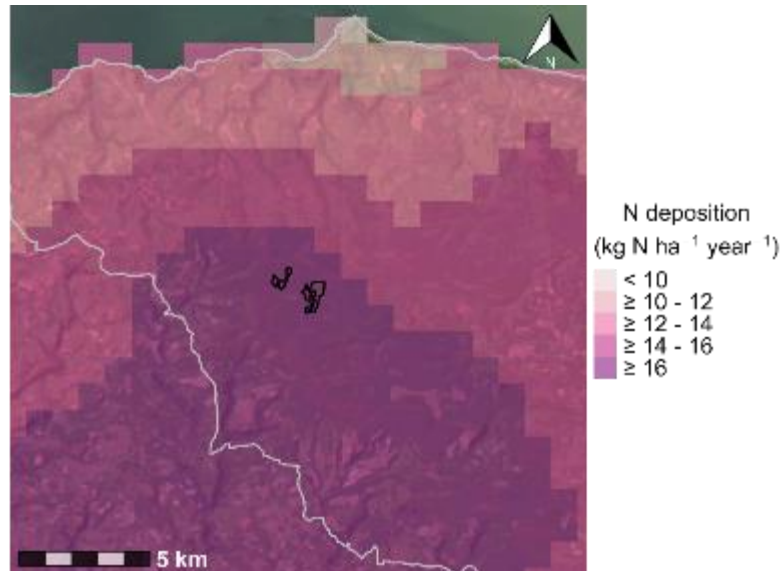


Figure 42: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for Pinkworthy and Driver Farm SSSI

Pinkworthy and Driver Farm SSSI has a varied range of sources of N deposition (**Figure 43**). The 2018 source attribution dataset estimates that the largest source of N deposition (to low-lying vegetation) received by Pinkworthy and Driver Farm is from livestock emission sources in England (~33%). However, within the National Park this is one of the smaller proportions of English livestock, compared with other designated sites. European emission sources contribute almost a quarter of N deposition (~22%), followed by international shipping (12%).

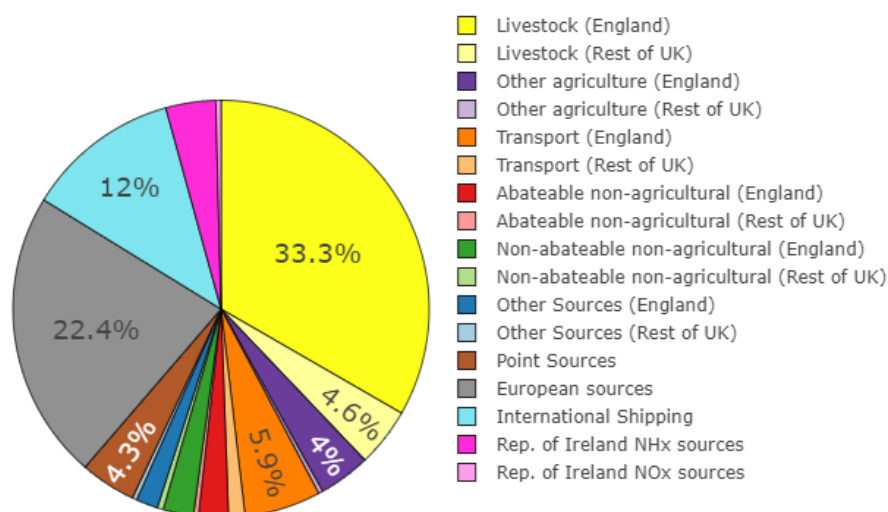


Figure 43: Source Attribution (2018) Total N deposition received by low-growing semi-natural vegetation for Pinkworthy and Driver Farm SSSI

4.10 Porlock Ridge and Saltmarsh SSSI

Porlock Ridge and Saltmarsh SSSI comprises a shingle ridge and the associated saltmarsh hinterland, extending for a distance of approximately 4 km along the west Somerset coast, immediately north of Porlock village. This site is notified for its nationally important active coastal geomorphological features¹⁵. It is also nationally important for the saltmarsh and coastal vegetated shingle habitats which occur here. According to APIS, the site has been assigned a critical level of 1 $\mu\text{g NH}_3 \text{ m}^{-3}$ for its grassland features.

Table 9 Critical Loads and levels for features of Porlock Ridge and Saltmarsh SSSI

Feature Name	Are Bryophytes integral?	Are Lichens integral?	NH ₃ CLe ($\mu\text{g}/\text{m}^3$)	Minimum CL (KgN/ha/yr)	Maximum CL (KgN/ha/yr)
Sm11 - aster tripolium var. discoides - saltmarsh	No	No	1 or 3	10	20
Transitional Low Marsh Vegetation With Puccinellia Maritima, Annual Salicornia Species And Suaeda Maritima.	No	No	1 or 3	10	20
Suaeda Maritima Saltmarsh	No	No	1 or 3	10	20
Rumex Crispus - Glaucium Flavum Shingle Community	No	No	1 or 3		
Annual Salicornia Saltmarsh	No	No	1 or 3	10	20

¹⁵ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/2000380.pdf>

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Festuca Rubra - Armeria Maritima Maritime Grassland	No	No	1 or 3		
Festuca Ovina - Agrostis Capillaris - Rumex Acetosella Grassland	Yes	Yes	1	5	15
Atriplex Portulacoides Saltmarsh	No	No	1 or 3	10	20
Aster Tripolium Var. Discoides Saltmarsh	No	No	1 or 3	10	20

The site is located in the less agriculturally intensive north of the National Park, with no clear emission hotspots nearby according to the NAEI emission maps (Figure 44). Agricultural emission density generally increases with distance from the site and beef and sheep farming are key emission sources. Ammonia concentrations for all of the SSSI are modelled to be below $1 \mu\text{g NH}_3 \text{ m}^{-3}$, (Figure 45) with an area-weighted average NH_3 concentration estimated at $0.8 \mu\text{g NH}_3 \text{ m}^{-3}$.

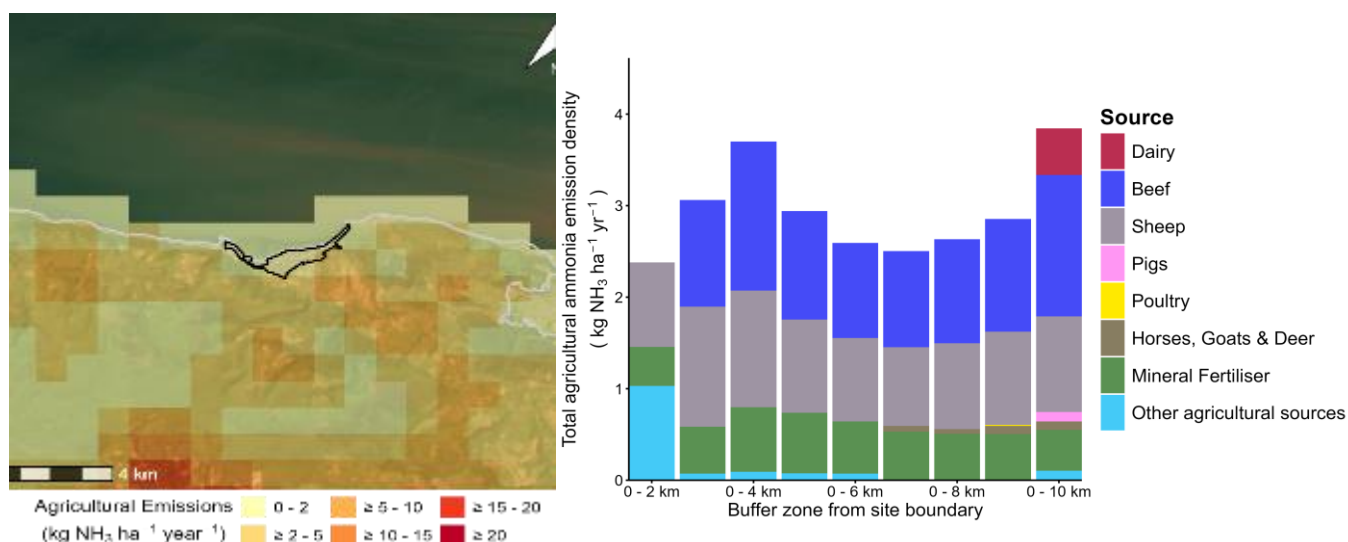


Figure 44: Gridded 2021 agricultural emission estimates (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for Porlock and Saltmarsh SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).



Figure 45: Mean modelled NH₃ concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Porlock Ridge and Saltmarsh SSSI

Only nitrogen deposition estimates onto low-growing semi-natural vegetation is described for N deposition, as this matches the vegetation types in the SSSI. The area-weighted mean rate of N deposition to low-growing vegetation (where present) to the site is relatively low at ~10 kg N ha⁻¹ yr⁻¹, with ~ 51% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, **Figure 46**).

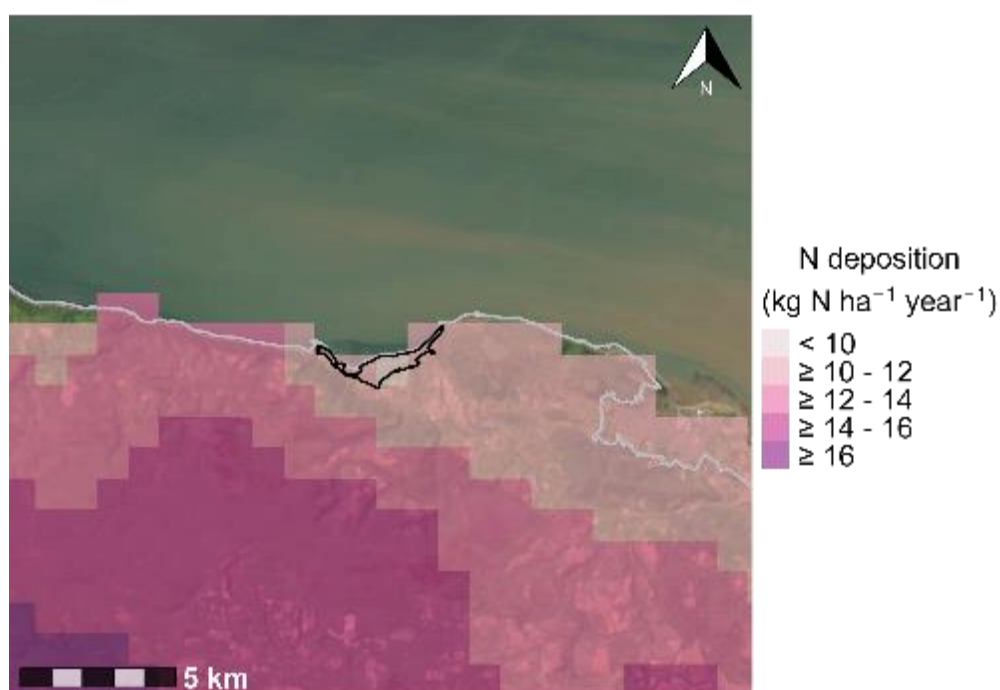


Figure 46: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for Porlock Ridge and Saltmarsh SSSI

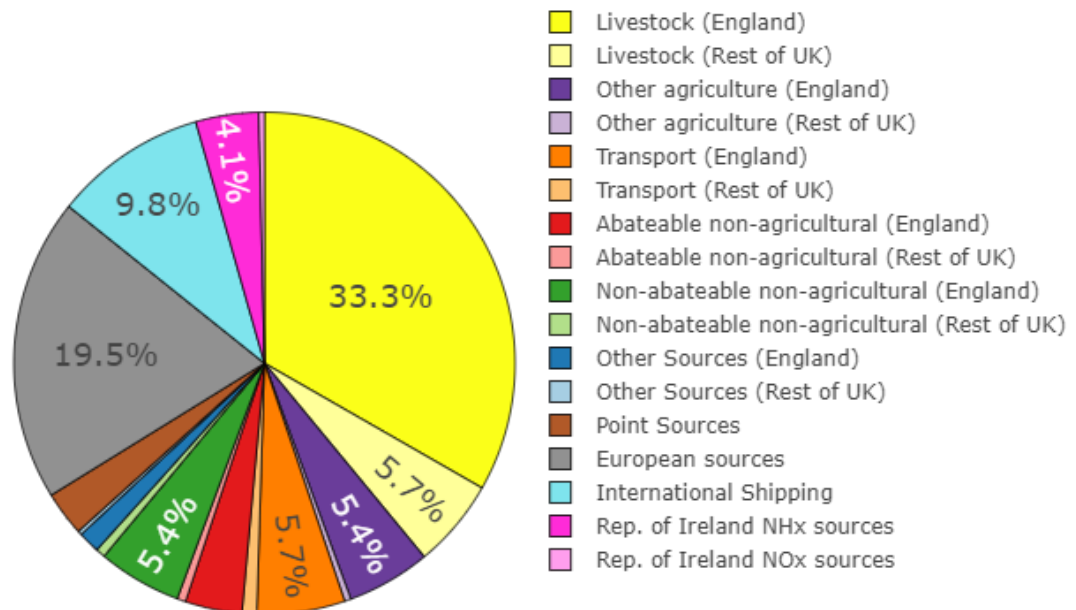


Figure 47: Source Attribution (2018) Total N deposition received by low-growing semi-natural vegetation for Porlock Ridge and Saltmarsh SSSI

Similarly to other more coastal sites, Porlock Ridge and Saltmarsh SSSI has a much more varied range of sources of N deposition than other sites within the National Park (**Figure 47**). The 2018 source attribution dataset estimates that a third of N deposition (to low-lying vegetation) received by Porlock Ridge is from livestock emission sources in England (~33%). Livestock emission sources from other UK countries, largely from Wales given the locations of the site, contributes to a higher proportion of N deposition than at other sites in Exmoor NP (~6%). Together, international shipping and European emission sources contribute to almost 30% of total N deposition, with international shipping having a larger influence on N deposition given its proximity to the Bristol Channel.

4.11 River Barle SSSI

The River Barle SSSI is a large river system spanning across the south and west of Exmoor National Park. The Barle is a river of very high water quality with limited influence of human activity¹⁶. The upper reaches are oligotrophic, flowing off moorland. The site has been assigned a critical level of 1 or 3 $\mu\text{g NH}_3 \text{ m}^{-3}$.

¹⁶ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/2000143.pdf>

Table 10 Critical Loads and levels for features of River Barle SSSI

Feature Name	Any Bryophytes integral?	Are Lichens Integral?	NH ₃ CLe (µg/m ³)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
Flowing waters - Type IX: oligotrophic, mountains and moorland	No	No	1 or 3		
Flowing waters - Type X: fast-flowing, oligotrophic, base-poor rivers	No	No	1 or 3		
Molinia Caerulea - Potentilla Erecta Mire	Yes	No	1 or 3	15	25
Atlantic Salmon			Not sensitive		
Juncus Effusus / Acutiflorus - Galium Palustre Rush Pasture	No	No	3	15	25
Filipendula Ulmaria - Angelica Sylvestris Mire	No	No	3	15	25
Cynosurus Cristatus - Centaurea Nigra Grassland	No	No	3	10	20
Cynosurus Cristatus - Centaurea Nigra Grassland	No	No	3	10	20
Flowing waters - Type VII: low-gradient streams at a relatively high altitude/latitude, flowing over a moderately basic geology	No	No	1 or 3		
Flowing waters - Type VIII: rivers common throughout western Britain over hard rocks	No	No	1 or 3		

Agricultural emissions within the vicinity of the river Barle are relatively low, and increases in the more southeastern areas of the SSSI (Figure 48). Agricultural emission density generally increases with distance from the site and beef farming is a key emission source. Ammonia concentrations for the SSSI are modelled to be below 1 µg NH₃ m⁻³ in the more northern areas and increase to between 1 and 1.5 µg NH₃ m⁻³ in the southern region of the national park (Figure 49). The whole site has an estimated area-weighted average NH₃ concentration of 0.95 µg NH₃ m⁻³.

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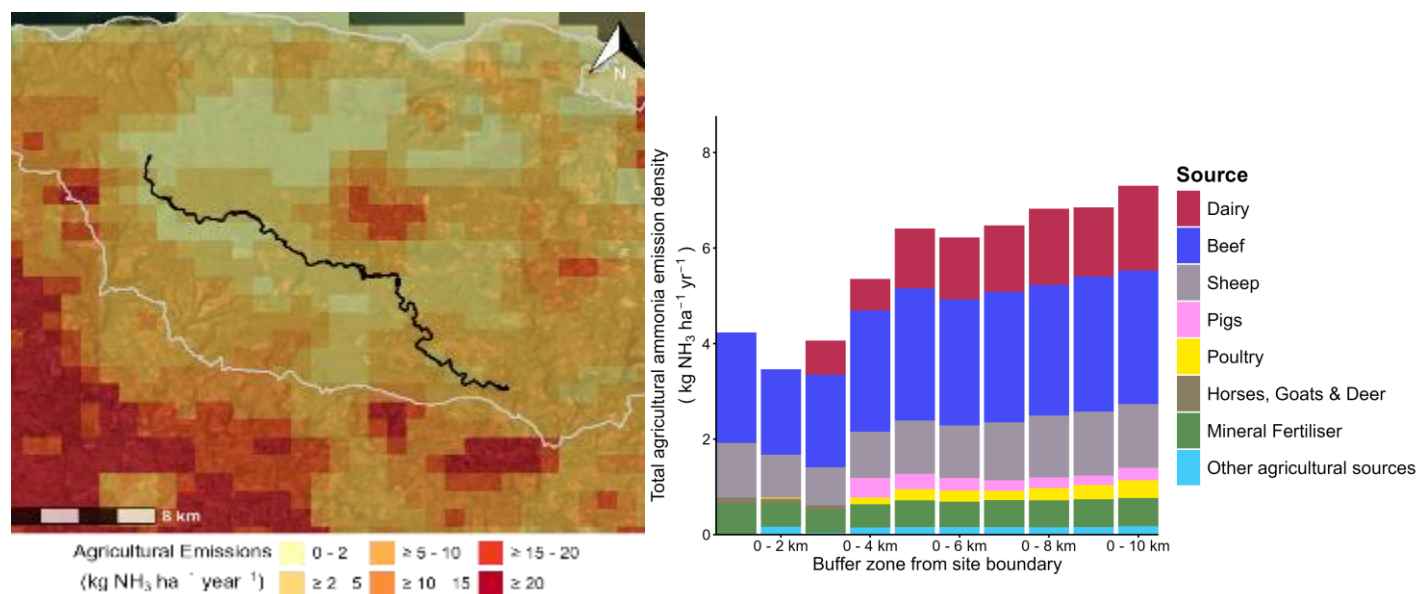


Figure 48: Gridded 2021 agricultural emission estimates for (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for River Barle SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

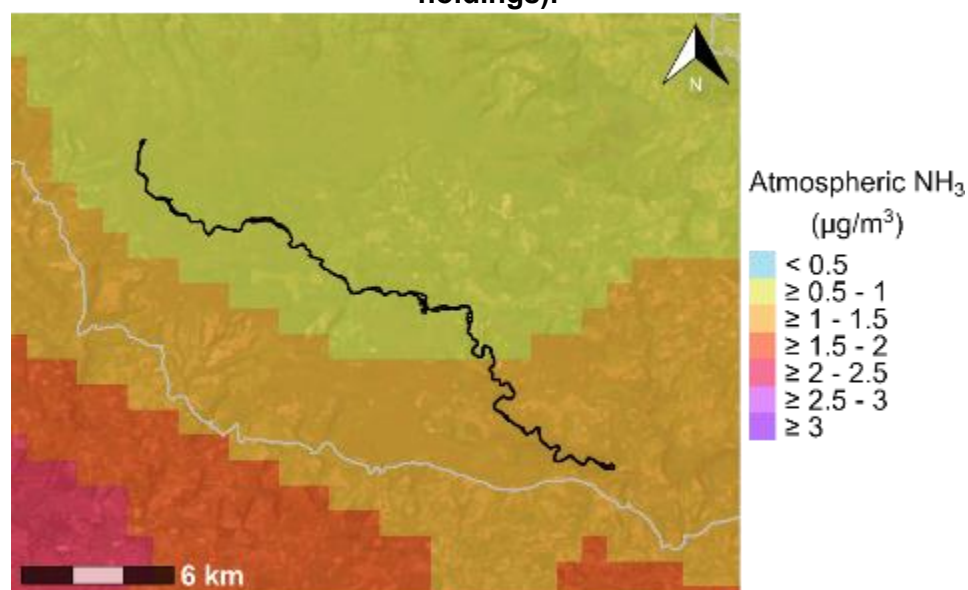


Figure 49: Mean modelled NH₃ concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for River Barle SSSI

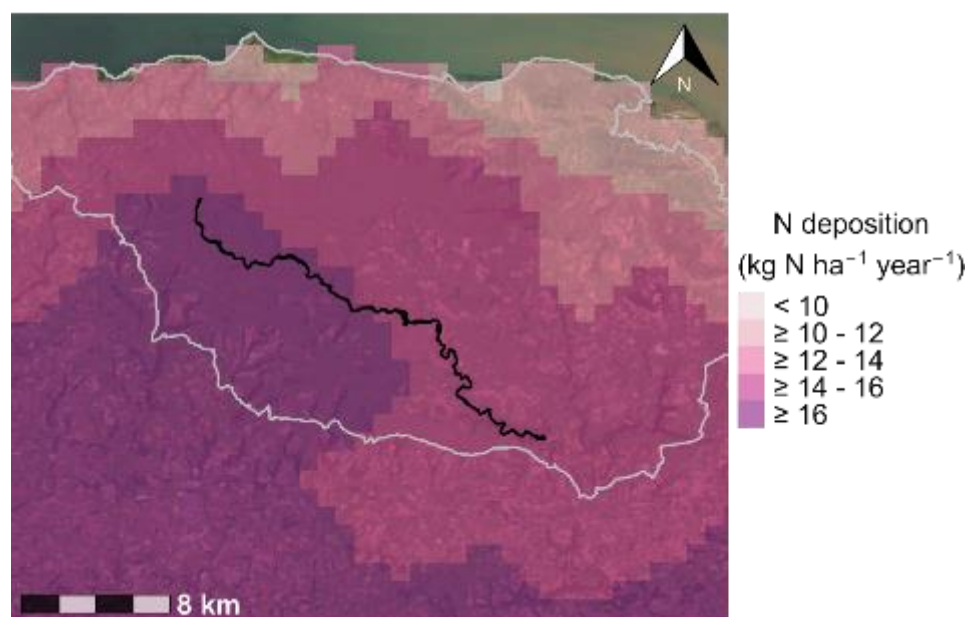


Figure 50: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for River Barle

Only nitrogen deposition estimates onto low-growing semi-natural vegetation will be described for N deposition. The area-weighted mean rate of N deposition to low-growing vegetation (where present) to the site is 15.8 kg N ha⁻¹ yr⁻¹, with ~ 50% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, **Figure 50**).

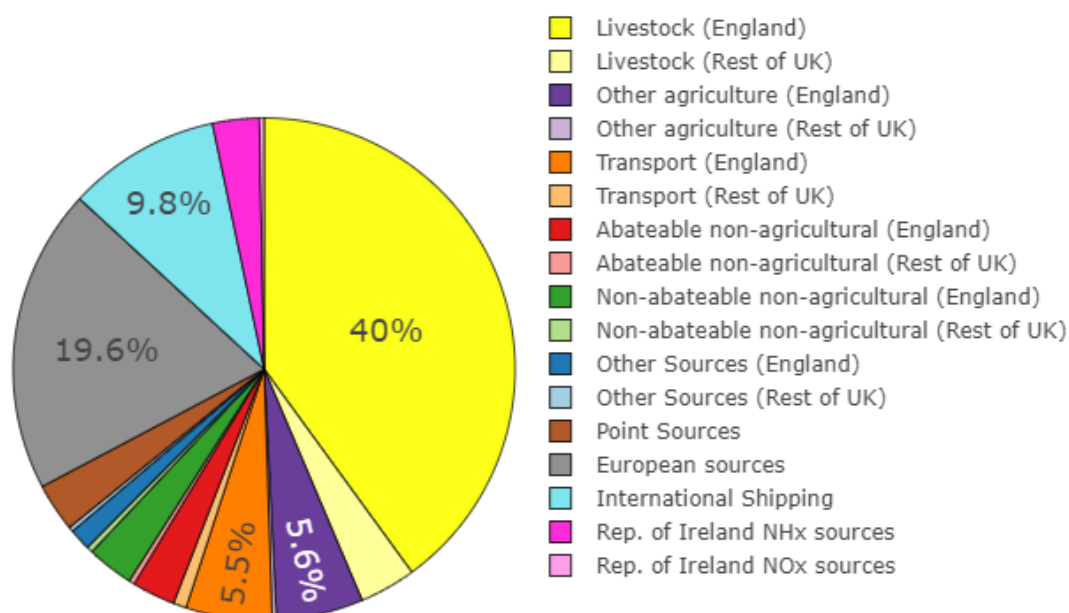


Figure 51: Source Attribution (2018) Total N deposition received by low-growing semi-natural vegetation for River Barle SSSI.

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The 2018 source attribution dataset (FRAME 2018) estimates that a third of nitrogen deposition (to low-lying vegetation) received by the Barle is livestock emission sources in England (~40%). Livestock sources from across the rest of the UK contribute a relatively small proportion and given the location of the site, these livestock sources from the rest of the UK are likely to primarily originate from sources within Wales. Transport contributes ~5.5% of the total N deposition received.

A substantial proportion of nitrogen deposition is also estimated to originate from sources outside the UK, with European sources contributing almost a fifth (~19.6%), international shipping ~9.8%. These sources may require international intervention to reduce their contribution to N deposition received by the site.

4.12 River Lyn SSSI

The River Lyn within a catchment consisting of moorland on the west of Exmoor. The river was the scene of a unique rainfall sequence and flood event in 1952 which created a number of unique geomorphological landforms still visible today¹⁷. The SSSI status is for these unique features and therefore the site has no associated critical level.

The NAEI agricultural ammonia emissions data suggest agricultural emissions within the site staying below 10 kg NH₃ ha⁻¹ (**Figure 52**). Agricultural emission sources in the surrounding areas are predominantly beef.

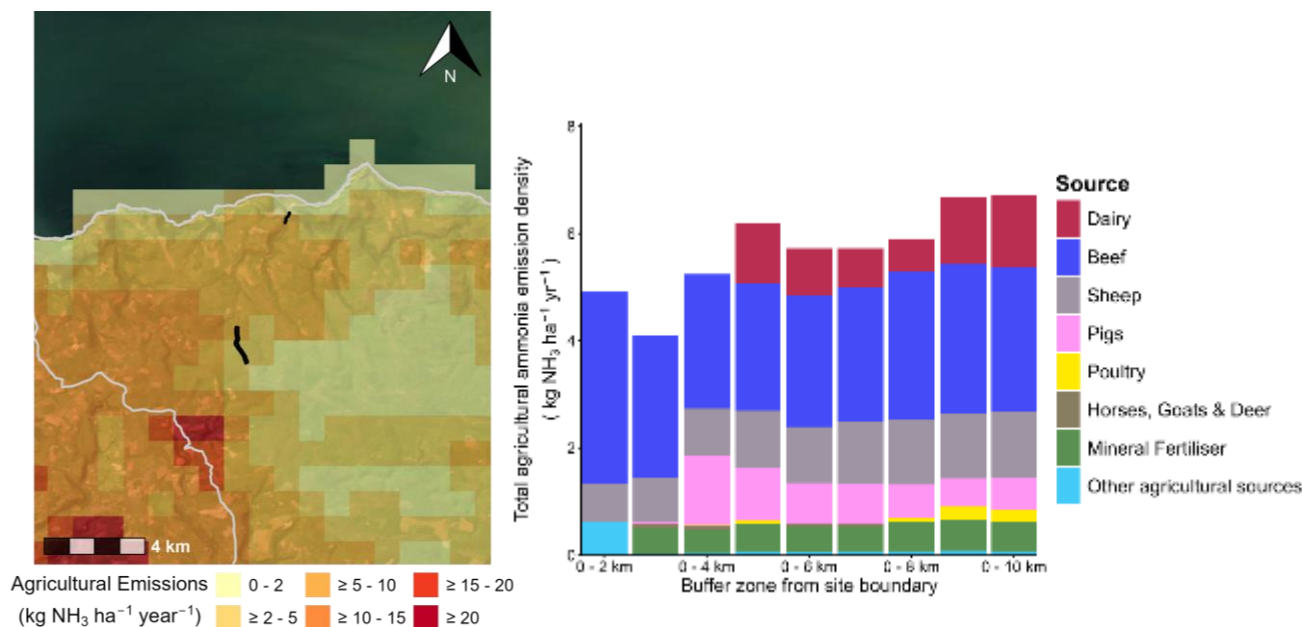


Figure 52: Gridded 2021 agricultural emission estimates (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for River Lyn for

¹⁷ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/2000088.pdf>

River Lyn SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

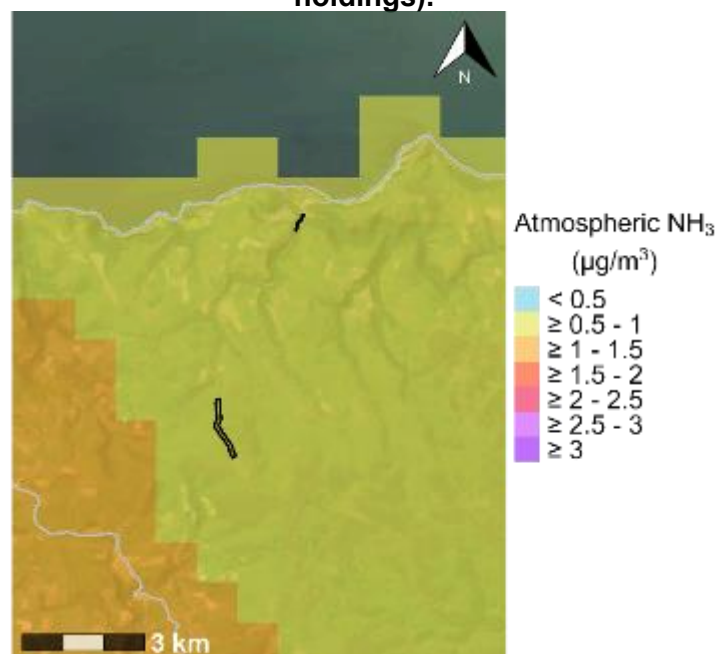


Figure 53: Mean modelled NH₃ concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for River Lyn SSSI

All of the SSSI is modelled to have an ammonia concentration below 1 µg NH₃ m⁻³ (**Figure 53**) and overall, the estimated area-weighted average NH₃ concentration is 0.9 µg NH₃ m⁻³.

The area-weighted mean rate of N deposition to low-growing vegetation (where present) to the site is 15.2 kg N ha⁻¹ yr⁻¹, with ~ 48% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, **Figure 54**).

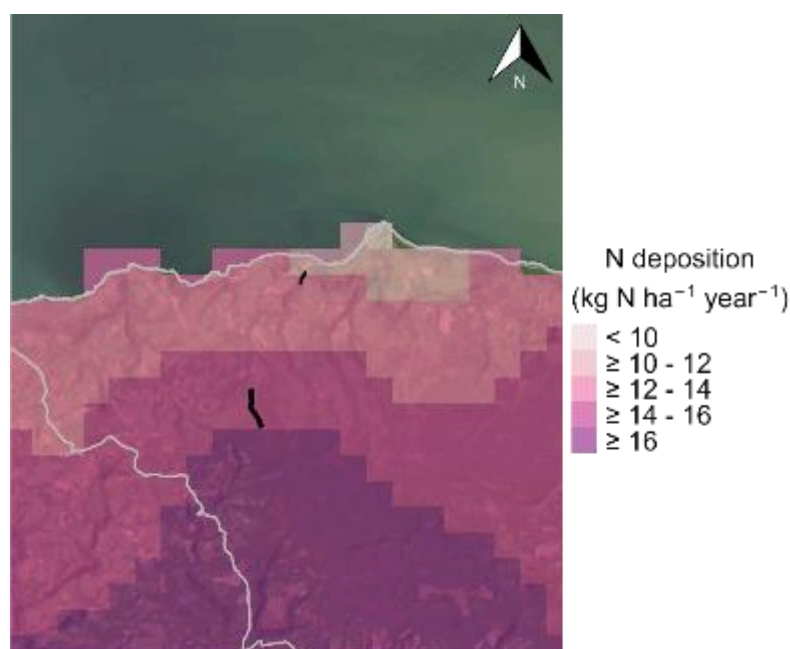


Figure 54: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for River Lyn

4.13 South Exmoor SSSI

South Exmoor SSSI contains an extensive area of heathlands unique to this region of England and Wales, as well as transitions between different heathland and semi-natural scrubs and woodlands¹⁸. The most sensitive critical level for the site is 1 µg for dry and wet heath species. Almost all of this site is within the Exmoor Heaths SAC.

Table 11 Critical Loads and levels for features of South Exmoor SSSI

Feature Name	Any Bryophytes integral?	Are Lichens Integral?	NH ₃ Cle (µg/m ³)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
Calluna Vulgaris - Erica Cinerea Heath	Yes	Yes	1	5	15
Calluna Vulgaris - Ulex Gallii Heath	Yes	No	1 or 3	5	15
Juncus Effusus / Acutiflorus - Galium Palustre Rush Pasture	No	No	3	15	25
Calluna Vulgaris - Vaccinium Myrtillus Heath	Yes	Yes	1	5	15
Carex Echinata - Sphagnum Recurvum (Fallax) /Auriculatum (Denticulatum) Mire	Yes	No	1	5	15

¹⁸ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1002269.pdf>

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Vaccinium Myrtillus - Deschampsia Flexuosa Heath	Yes	No	1	5	15
Hypericum Elodes - Potamogeton Polygonifolius Soakway	Yes	No	1	5	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Scirpus Cespitosus - Erica Tetralix Wet Heath	Yes	Yes	1	5	15
Quercus Spp.-Betula Spp.-Deschampsia Flexuosa Woodland	Yes	Yes	1 or 3	10	15
Erica Tetralix - Sphagnum Compactum Wet Heath	Yes	Yes	1	5	15
Molinia Caerulea - Potentilla Erecta Mire	Yes	No	1 or 3	15	25
Quercus Petraea - Betula Pubescens - Oxalis Acetosella Woodland	Yes	Yes	1 or 3	10	15

South Exmoor is on the southern National Park boundary and within 1km of an agricultural NH₃ emission hotspot with emissions >20 kg NH₃ ha⁻¹ (Figure 55). Agricultural emission density generally increases with distance from the site, with dairy becoming a notable emission source within 2km of the site boundary (Figure 53). Key emission sources within 1km of the site are beef and sheep.

Ammonia concentrations for most of South Exmoor are modelled to be between 1 and 1.5 µg NH₃ m⁻³ (Figure 57), with an area-weighted average NH₃ concentrations are estimated at 1.1 µg NH₃ m⁻³. This is above the critical level for which has been assigned to the site. With higher ammonia concentrations and local emission sources, this site has potential to benefit from some targeted measures. There appears to be a north-south gradient of ammonia concentrations at the site, with the northern area seeing modelled concentrations of below 1 µg NH₃ m⁻³. This is in line with the agricultural emissions data showing hotspots to the south of the site and the general trend towards higher atmospheric nitrogen pollution in the areas immediately surrounding the Exmoor boundary.



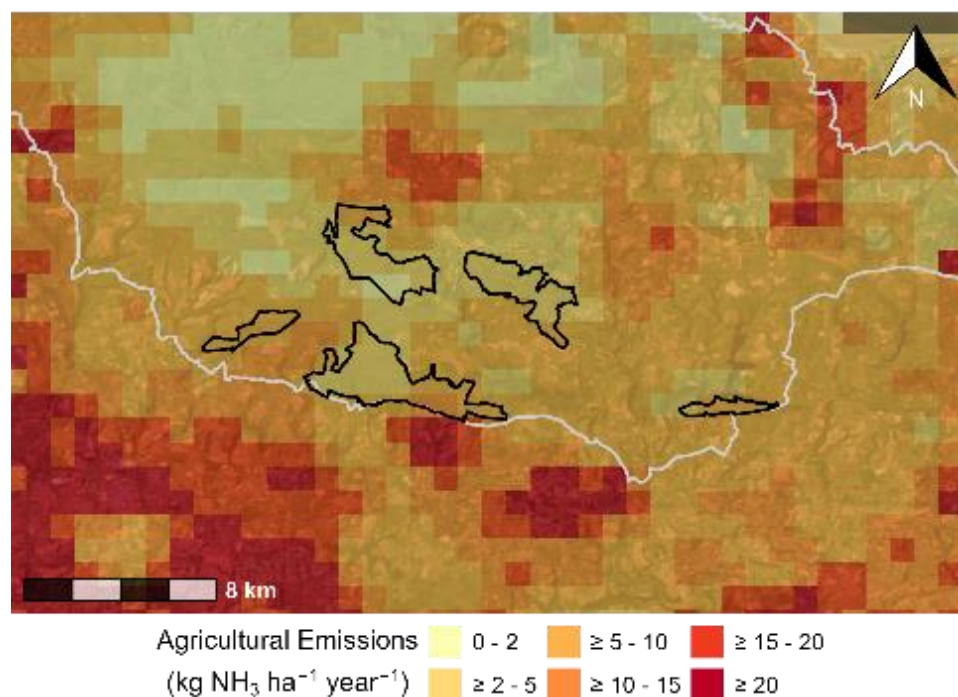


Figure 55: Gridded 2021 agricultural emission estimates for South Exmoor SSSI (1km x 1km grid resolution, NAEI)

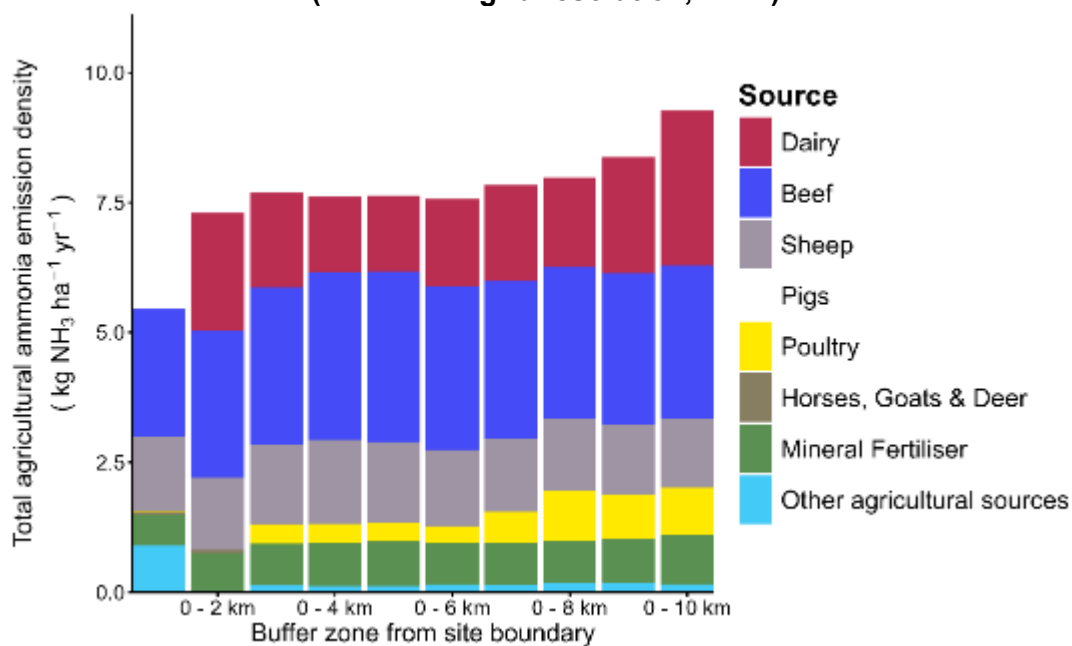


Figure 56: Agricultural emission density estimates for South Exmoor SSSI. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

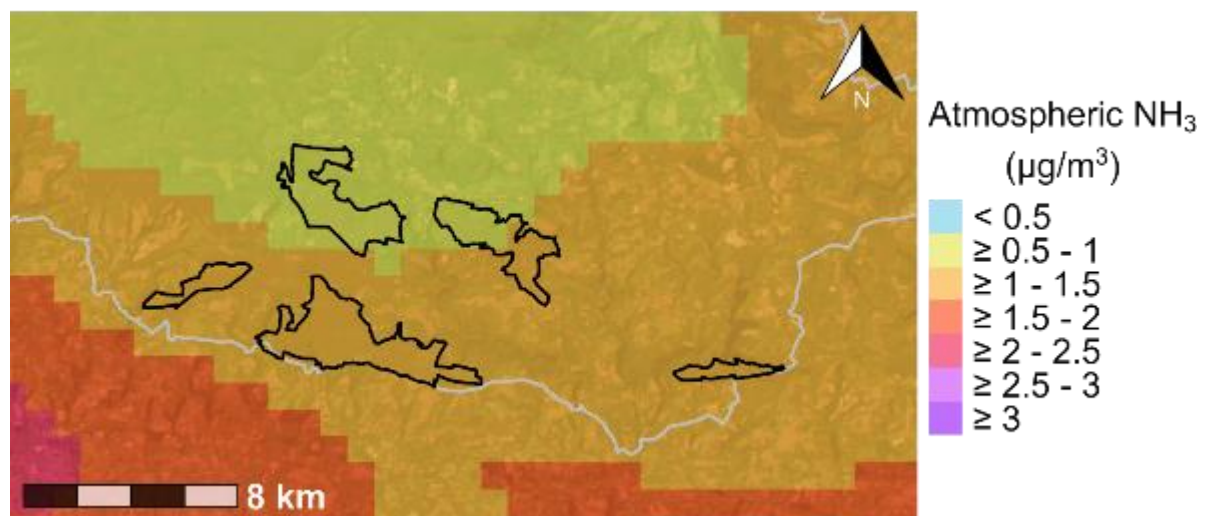


Figure 57: Mean modelled NH₃ concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for South Exmoor SSSI

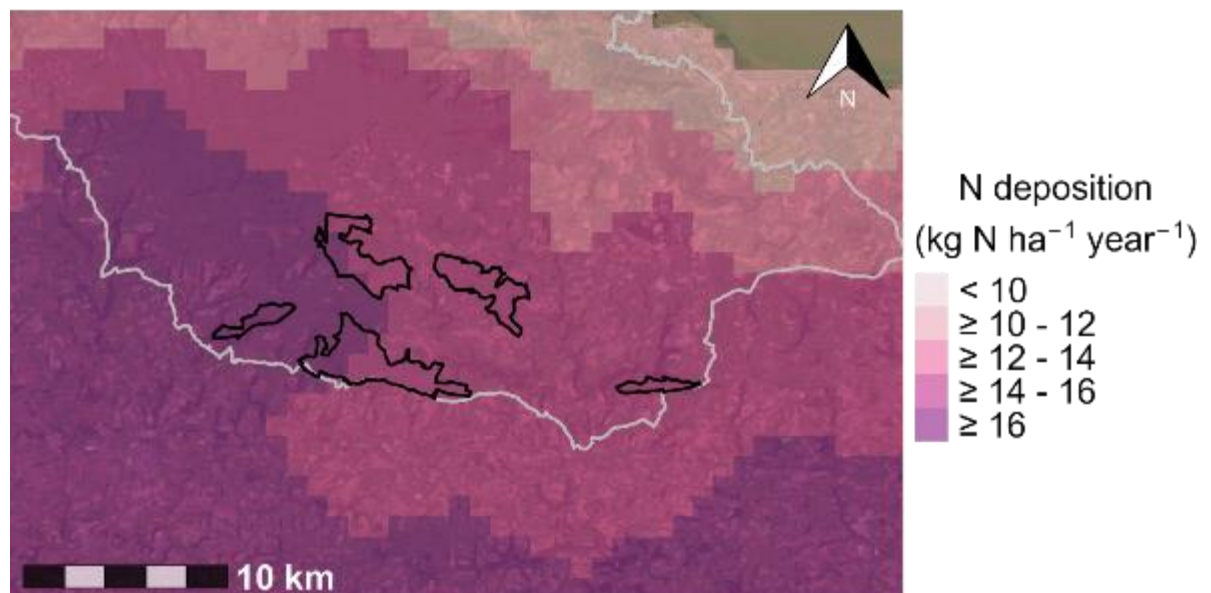


Figure 58: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for South Exmoor SSSI

Nitrogen deposition estimates onto low-growing semi-natural vegetation will be described in the following section, as this matches the heathland habitats in the SSSI. This also matches the Annex 1 heath habitats classified by the JNCC for the Exmoor Heaths SAC, which most of this site is a part of. The area-weighted mean rate of N deposition to low-lying vegetation (where present) to the site is 15.7 kg N ha⁻¹ yr⁻¹, with ~ 54.7 % of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 58).

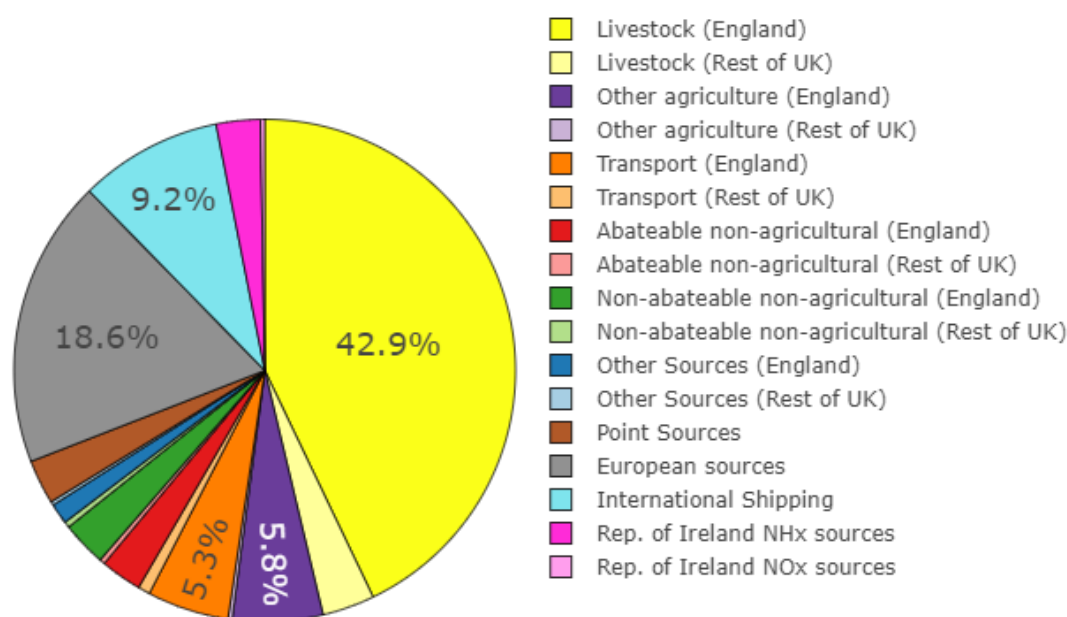


Figure 59: Source Attribution (2018) Total N deposition received by low-growing semi-natural vegetation for South Exmoor

The 2018 source attribution dataset (FRAME 2018) estimates that the largest source of nitrogen deposition (to low-growing semi-natural vegetation) received by South Exmoor is livestock emission sources in England (~42.9%, Figure 59). Transport contributes only ~5.3% of the total N deposition received.

A substantial proportion of nitrogen deposition is also estimated to originate from sources outside the UK, with European sources contributing ~18.6%, international shipping ~9.2%. These sources may require international intervention to reduce their contribution to N deposition received by the site.

4.14 Watersmeet SSSI

Watersmeet is an extensive area of ancient oak woodland located in a river valley system to the north of Exmoor, near the coastal town of Lynton. The site is one of the largest remaining semi-natural ancient woodlands in the South West. There are over 50 species of lichen present in the site, due to the moist conditions favourable to their growth¹⁹. Therefore, a most sensitive critical level of 1 µg has been assigned to the site. Almost all of Watersmeet is part of the Exmoor and Quantock Oakwoods SAC.

¹⁹ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1003886.pdf>

Table 12 Critical Loads and levels for features of Watersmeet SSSI

Feature Name	Ary Bryophytes integral?	Are Lichens Integral?	NH ₃ CL _e (µg/m ³)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Vascular Plant Assemblage			3		
Alnus Glutinosa - Fraxinus Excelsior - Lysimachia Nemorum Woodland	Yes	Yes	1 or 3		
Combinations Of Species - Lichens	No	Yes	1		
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Quercus Petraea - Betula Pubescens - Dicranum Majus Woodland	Yes	Yes	1 or 3	10	15
Quercus Petraea - Betula Pubescens - Oxalis Acetosella Woodland	Yes	Yes	1 or 3	10	15
Quercus Robur - Pteridium Aquilinum - Rubus Fruticosus Woodland	Yes	Yes	1 or 3	15	20
Quercus Spp.-Betula Spp.-Deschampsia Flexuosa Woodland	Yes	Yes	1 or 3	10	15

The NAEI agricultural ammonia emission estimates suggest the site is in an area of lower agricultural emissions (Figure 60). Agricultural emission density increases noticeably 0 - 2km compared to 0 - 1km from the site. This is mostly driven by an increasing in the emission density of beef farming.

Overall, the SSSI has an estimated area-weighted average NH₃ concentration of 0.8 µg NH₃ m⁻³, below the critical level for the most sensitive features on the site (1 µg). Modelled ammonia concentrations in the surrounding area also suggest an ammonia concentration below 1 µg (Figure 61).

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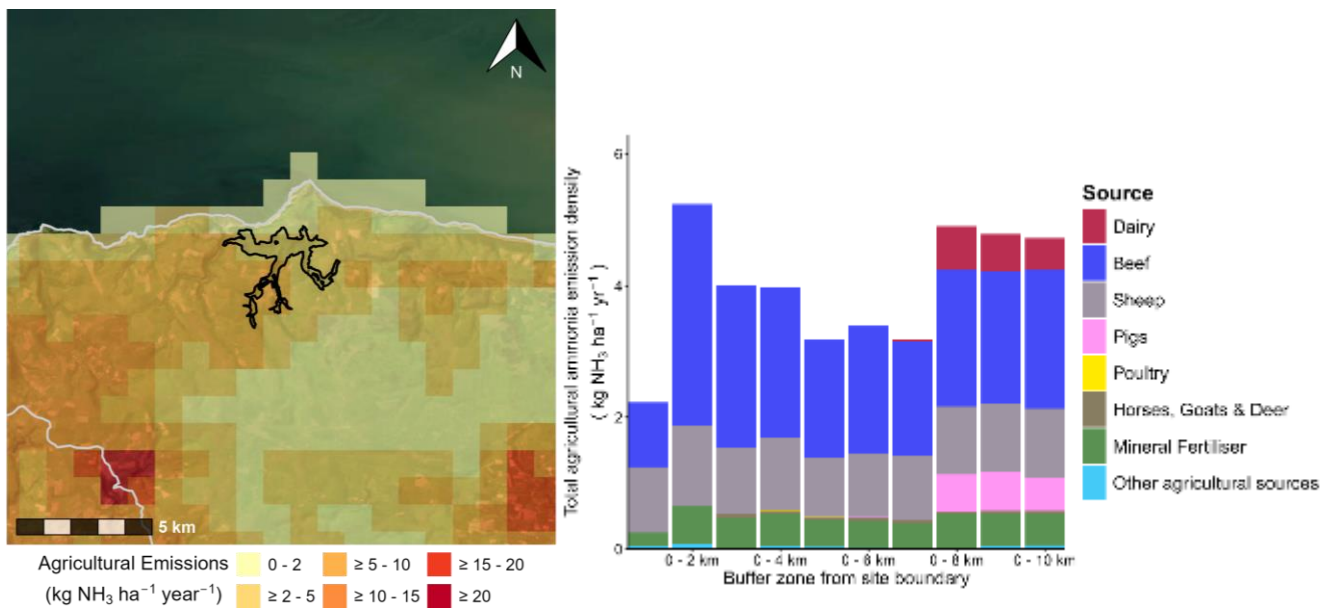


Figure 60: Gridded 2021 agricultural emission estimates (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for Watersmeet. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).



Figure 61: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Watersmeet SSSI

Only nitrogen deposition estimates onto woodland features will be described in the following section, as this is most suited to the woodland habitats in the SSSI. The area-weighted mean rate of N deposition to woodland features (where present) to the site is $18.7 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, with ~ 60.9% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 62).



Figure 62: CBED mean modelled N deposition for 2019 – 2021 onto woodland features, for Watersmeet

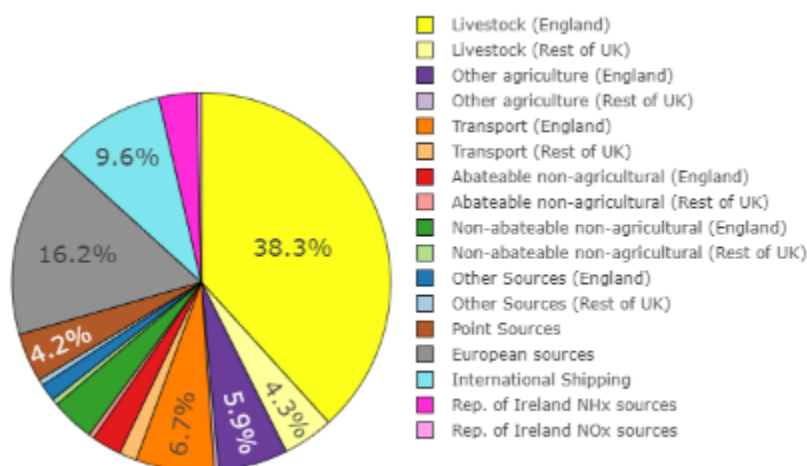


Figure 63: Source Attribution (2018) Total N deposition received by woodland features for Watersmeet

Similar to other coastal sites, Watersmeet has a more varied range of sources of nitrogen deposition than others within the national park (Figure 63). The 2018 source attribution dataset (FRAME 2018) estimates that the largest source of nitrogen deposition (to woodland vegetation) received by Watersmeet SSSI is livestock emission sources in England (~38.3%). Transport from England contributes ~6.7% of the total N deposition received.

A notable proportion of nitrogen deposition is estimated to originate from sources outside the UK, with European sources contributing ~16.2%, international shipping and ~10%.

4.15 West Exmoor Coast and Woods SSSI

West Exmoor Coast and Woods SSSI is a combination of heathlands towards the coast and ancient oak woodlands in the valleys. The site features a rich assemblage of over 100 species of coastal lichen flora, including several old-forest

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indicators²⁰. Due to the presence of dry heaths and lichens, a critical level of 1 µg has been assigned to this site. The SSSI area overlaps with both of the Exmoor SAC areas, the Exmoor Heaths SAC in the northern coastal heath area and Exmoor and Quantock Oakwoods SAC in the woodland valley areas.

Table 13: Critical Loads and levels for features of West Exmoor Coast and Woods SSSI

Feature Name	Any Bryophytes Integral?	Are Lichens Integral?	NH ₃ CL _e (µg/m ³)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
Festuca Rubra - Hyacinthoides Non-Scripta Maritime Bluebell Community	No	No	1 or 3		
Fraxinus Excelsior - Acer Campestre - Mercurialis Perennis Woodland	Yes	Yes	1 or 3	15	20
Quercus Petraea - Betula Pubescens - Oxalis Acetosella Woodland	Yes	Yes	1 or 3	10	15
Calluna Vulgaris - Vaccinium Myrtillus Heath	Yes	Yes	1	5	15
Combinations Of Species - Lichens	No	Yes	1		
Festuca Rubra - Armeria Maritima Maritime Grassland	No	No	1 or 3		
Calluna Vulgaris - Ulex Gallii Heath	Yes	No	1 or 3	5	15
Calluna Vulgaris - Erica Cinerea Heath	Yes	Yes	1	5	15
Quercus Robur - Pteridium Aquilinum - Rubus Fruticosus Woodland	Yes	Yes	1 or 3	15	20
Quercus Spp.-Betula Spp.-Deschampsia Flexuosa Woodland	Yes	Yes	1 or 3	10	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Ulex Gallii - Agrostis Curtisii Heath	Yes	Yes	1	5	15
Alnus Glutinosa - Fraxinus Excelsior - Lysimachia Nemorum Woodland	Yes	Yes	1 or 3		

In terms of agricultural NH₃ emissions, most the site is modelled to be in areas of relatively low emissions which gradually get higher in the more inland areas of the SSSI (Figure 64). Interestingly, the agricultural emission density is highest within the 0 – 1 km buffer zone of the site. However, there are too few farms to disclose which source type this is from within the 1km buffer zone area. Generally, Dairy has da

²⁰ <https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1004224.pdf>



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declining influence of agricultural emission density as buffer zones get further from the SSSI.

Overall, the SSSI has an estimated area-weighted average NH_3 concentration of $0.9 \mu\text{g NH}_3 \text{ m}^{-3}$, below the critical level for the most sensitive features on the site ($1 \mu\text{g}$). Although, ammonia concentrations follow a similar pattern to agricultural ammonia emissions, with the southernmost area of the SSSI estimated to have concentrations over $1 \mu\text{g m}^{-3}$ (Figure 65).

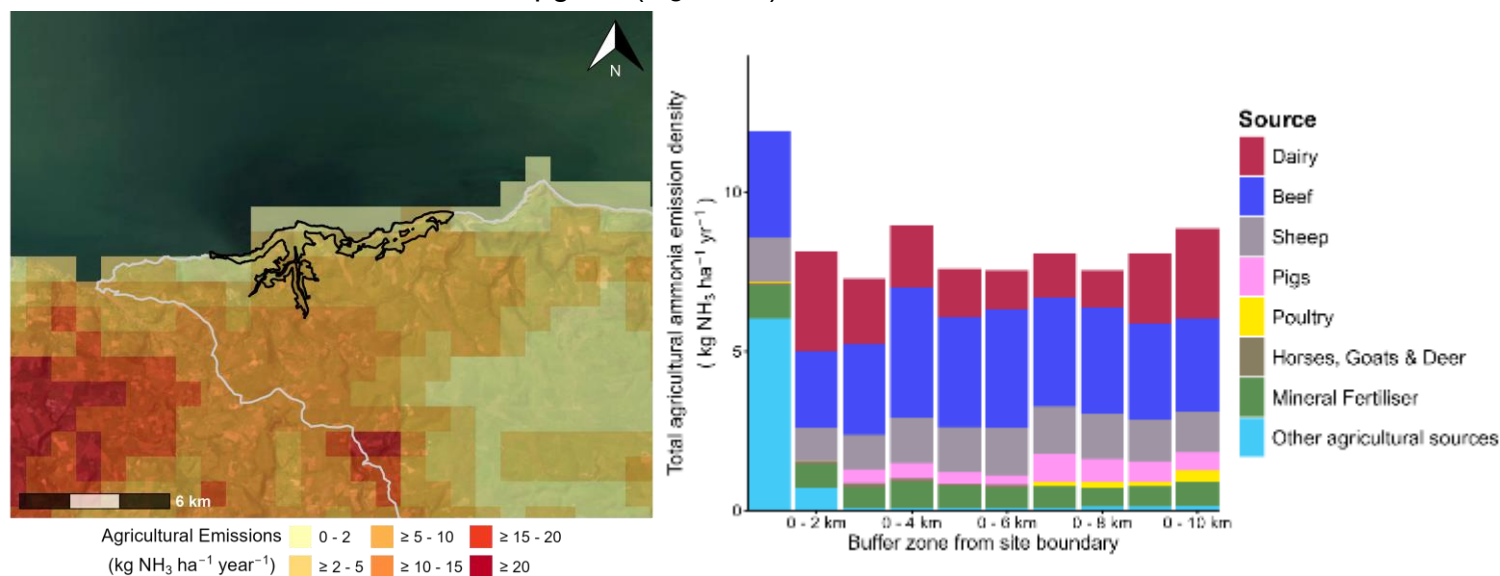


Figure 64: Gridded 2021 agricultural emission estimates (1km x 1km grid resolution, NAEI) (LEFT) and Agricultural emission density estimates (RIGHT) for West Exmoor Coast and Woods SSSI. The category ‘other agricultural sources’ refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).



Figure 65: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, West Exmoor Coast and Woods SSSI

As the site contains both ecologically important woodland features and low-lying vegetation features in the heathlands, both deposition types will be considered.

The area-weighted mean rate of N deposition to woodland vegetation (where present) to the site is relatively high at $20.1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, with ~ 66% of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 66). The 2018 source attribution dataset (FRAME 2018) estimates that the largest source of nitrogen deposition (to woodlands) received by the SSSI is livestock emission sources in England (~42%, Figure 67). Transport from England contributes ~6.2% of the total N deposition received. European sources account for a smaller proportion of total N deposition at this site compared to others within the National park, at 14.5%.



Figure 66: CBED mean modelled N deposition for 2019 – 2021 onto woodland features, for West Exmoor Coast and Woods SSSI

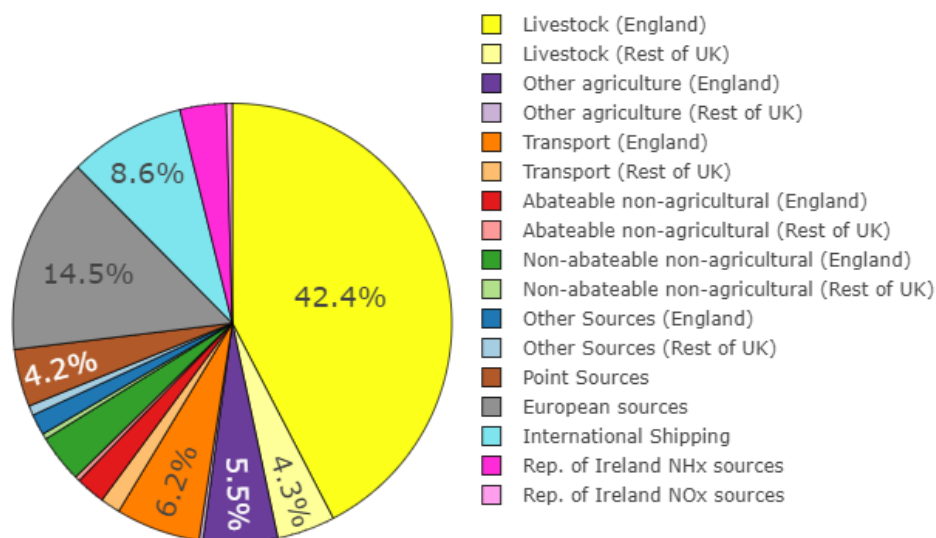


Figure 67: Source Attribution (2018) Total N deposition received by woodland features for West Exmoor Coast and Woods

The area-weighted mean rate of N deposition to low-lying vegetation (where present) to the site is relatively high at $\sim 13 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, with $\sim 53\%$ of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 68). The 2018 source attribution dataset (FRAME 2018) estimates that the largest source of nitrogen deposition (to woodlands) received by the SSSI is livestock emission sources in England ($\sim 41\%$, Figure 69). Transport from England contributes $\sim 5.2\%$ of the total N deposition received. In terms of nitrogen deposition estimated to originate from sources outside the UK, European sources contributing the second largest proportion ($\sim 14.5\%$), followed by international shipping ($\sim 8.6\%$).

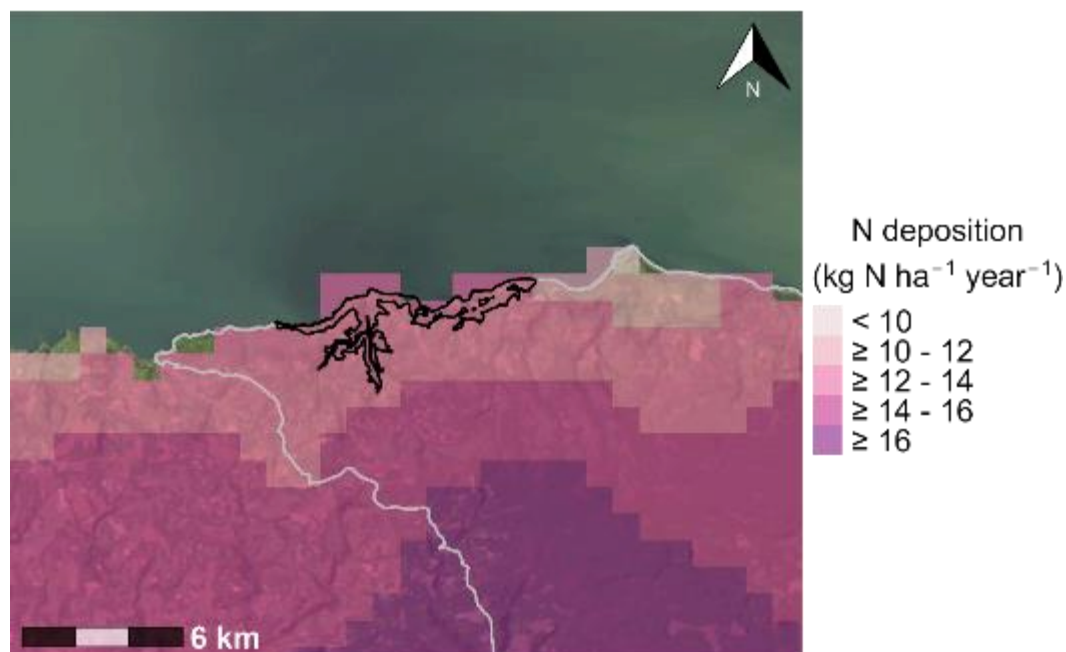


Figure 68: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for West Exmoor Coast and Woods

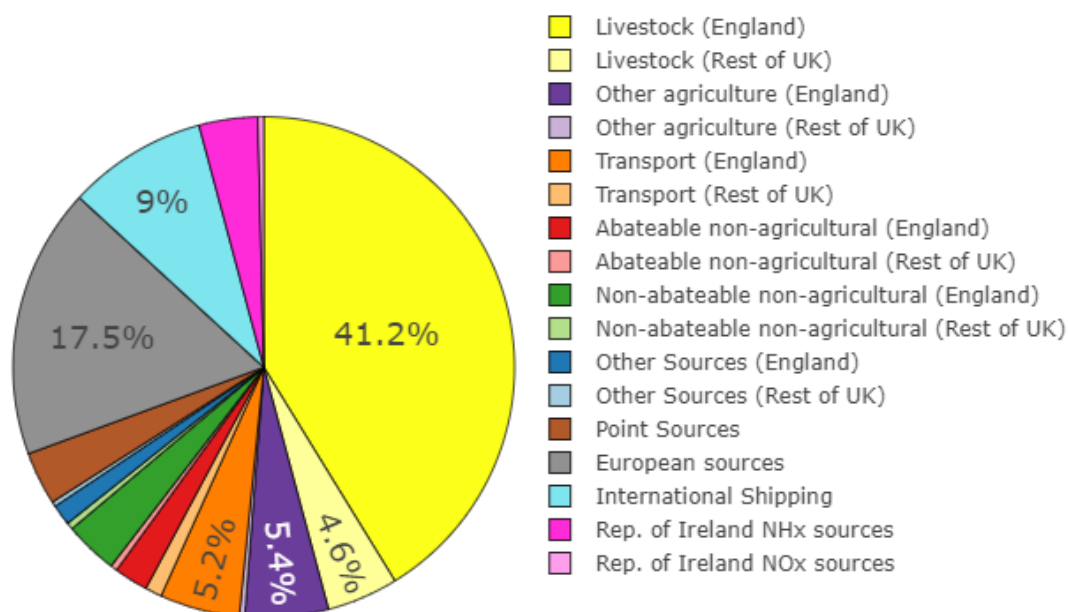


Figure 69: Source Attribution (2018) Total N deposition received by low-lying vegetation for West Exmoor Coast and Woods

4.16 Exmoor and Quantock Oakwoods SAC

Exmoor and Quantock Oakwoods SAC is a series of woodland areas spanning across Exmoor National Park. The SAC is consist of areas within 5 SSSI sites: Barle Valley SSSI, North Exmoor SSSI, The Quantocks SSSI (which is outside of Exmoor National Park), Watersmeet SSSI, West Exmoor Coast and Woods SSSI. The Annex I habitats which are the primary reason for designation of this site are old sessile oak woods with Ilex and Blechnum in the British Isles²¹. The sites are also home to otters, a number of bat species and are rich in bryophytes and epiphytic lichens. The site has been assigned a critical level of $1\mu\text{g NH}_3 \text{ m}^{-3}$. The most sensitive critical load for western acidic oak woodland is 10kg N ha.

Table 14 Critical Loads and levels for features of Exmoor and Quantock Oakwoods SAC

Feature Name	Are Bryophytes integral?	Are Lichens Integral?	NH ₃ CLe ($\mu\text{g}/\text{m}^3$)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
Alluvial forests with <i>Alnus glutinosa</i> and <i>Fraxinus excelsior</i> (Alno-Padion, Alnion incanae, Salicion albae)	Yes	Yes	1		
<i>Lutra lutra</i>			3		
<i>Cottus gobio</i>			Not sensitive		
<i>Lampetra planeri</i>			Not sensitive		
<i>Salmo salar</i>			Not sensitive		
European dry heaths	Yes	Yes	1	5	15
Northern Atlantic wet heaths with <i>Erica tetralix</i>	Yes	Yes	1	5	15
Blanket bogs (* if active bog)	Yes	Yes	1	5	10
Transition mires and quaking bogs	Yes	No	1	5	15
Old sessile oak woods with Ilex and Blechnum in the British Isles	Yes	Yes	1 or 3	10	15
Alkaline fens	Yes	No	1	15	25
Vegetated sea cliffs of the Atlantic and Baltic Coasts	No	No	1 or 3		

²¹ [Exmoor and Quantock Oakwoods - Special Areas of Conservation](#)

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Covering almost 1,900 ha, the woodland areas of this SAC are in differing areas of agricultural intensity. In areas of higher modelled agricultural intensity are within Barle valley and West Exmoor Coast and Woods SSSI's, while some areas of the SAC are within the upland heaths of North Exmoor SSSI and have a much lower intensity (Figure 70). Within a 2km buffer area of the SAC, the dominant agricultural emission sources are modelled to be beef and sheep (Figure 71)

Most of the SAC are modelled to experience concentrations below $1 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$, the critical level for lichens and bryophyte species present on the site (Figure 72). The areas of the SAC towards the south of the National Park are on the border of being above this threshold. Overall, the estimated area-weighted average NH_3 concentration is $1 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$, ranging from 0.6 to $1.7 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$.

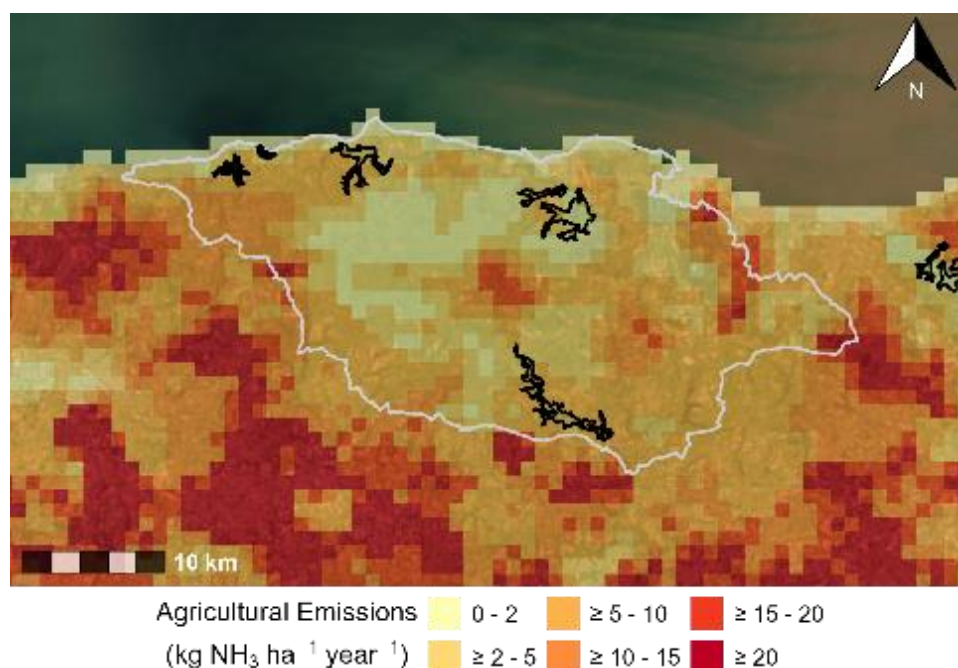


Figure 70: Gridded 2021 agricultural emission estimates for Exmoor and Quantock Oakwoods SAC (1km x 1km grid resolution, NAEI)

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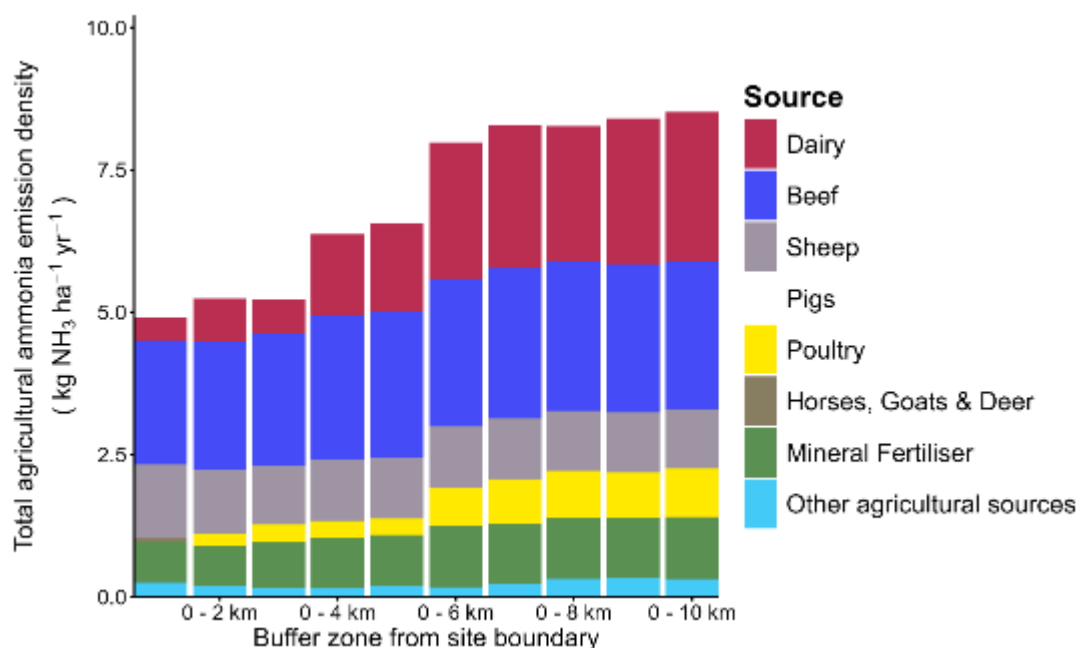


Figure 71: Agricultural emission density estimates for Exmoor & Quantock Oakwoods SAC. The category 'other agricultural sources' refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

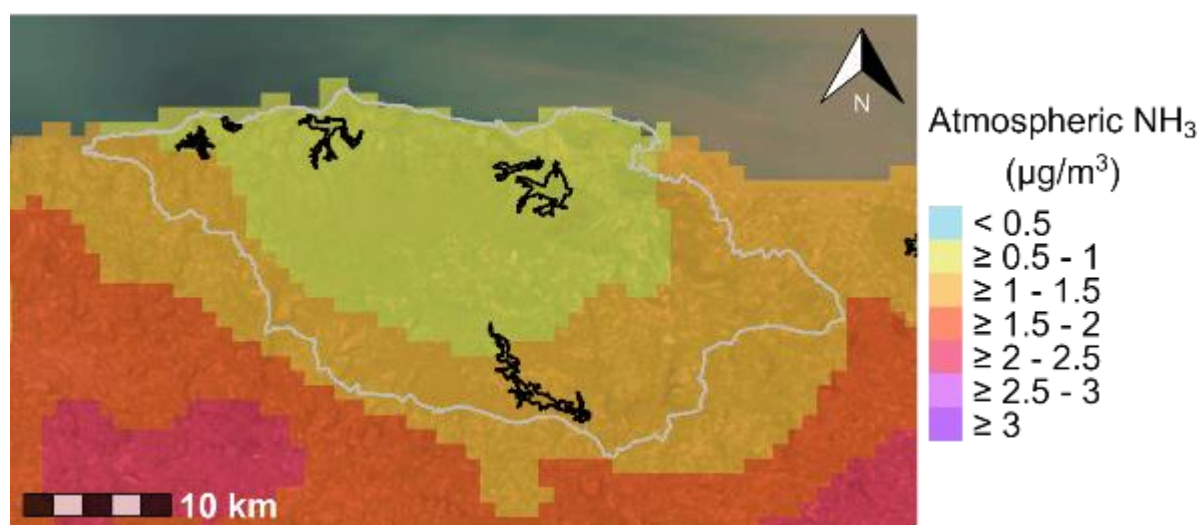


Figure 72: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Exmoor & Quantock Oakwoods SAC



Figure 73: CBED mean modelled N deposition for 2019 – 2021 onto woodland features, for Exmoor & Quantock Oakwoods SAC

Only nitrogen deposition estimates onto woodland features are described in the following section, as this matches the matches the Annex 1 Old sessile oak woods habitat classified by the JNCC for the Exmoor and Quantock Oakwoods SAC. The area-weighted mean rate of N deposition to woodland vegetation (where present) to the site is estimated at $21.2 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, with ~ 66 % of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 73).

The 2018 source attribution dataset estimates that the largest contribution to nitrogen deposition (to woodland features) received by Exmoor and Quantock Oakwoods SAC is from livestock emission sources in England (~44%, Figure 74). Livestock sources from across the rest of the UK contribute a relatively small proportion and given the location of the site, these livestock sources from the rest of the UK are likely to primarily originate from sources within Wales. Transport contributes ~6% of the total N deposition received. A substantial proportion of nitrogen deposition is also estimated to originate from sources outside the UK, with European sources contributing ~14%, and international shipping ~8%. These sources would require international intervention to reduce their contribution to N deposition received by the site.

Of the reduced nitrogen (NH_x) deposition, which is primarily from agricultural sources, two thirds of dry deposition from more local sources comes from livestock in England. Followed by just under 10% from other agriculture.

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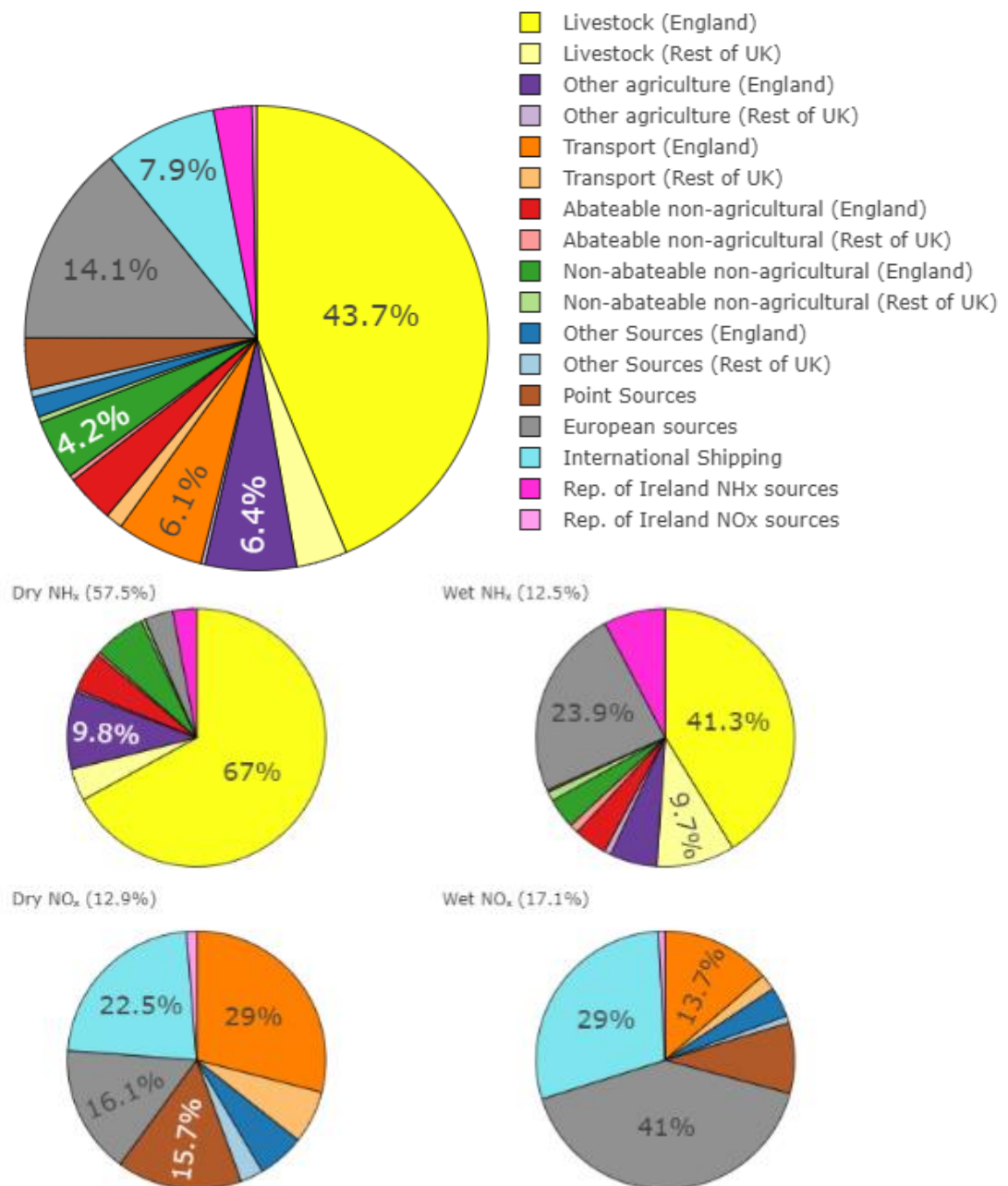


Figure 74: Source Attribution (2018) N deposition received by woodland-features for Exmoor & Quantock Oakwoods SAC. Top pie chart described total N Deposition, which is then broken down into its constituent parts in the four charts below.

4.17 Exmoor Heaths SAC

Exmoor Heaths SAC mainly consists of dry and wet heath habitats, with areas of blanket bog and some old sessile oak woodland. The wet heathland habitats in the protected area are highly variable through a combination of natural processes and modification through management, particularly burning. The Annex I habitats which are the primary reason for designation of this site are Northern Atlantic wet heaths, European dry heaths and *Erica tetralix*²². The SAC is consist of areas within 4 SSSI sites: Exmoor Coastal Heaths SSSI, North Exmoor SSSI, South Exmoor SSSI and West Exmoor Coast and Woods SSSI. The site has been assigned a critical level of $1 \mu\text{g NH}_3 \text{ m}^{-3}$, as bryophytes and lichens are considered important components of the heathlands. The most sensitive critical load is 5 kg N ha for blanket bogs, dry heaths wet heath quaking bogs.

Table 15 Critical Loads and levels for features of Exmoor Heaths SAC

Feature Name	Are Bryophytes Integral?	Are Lichens Integral?	NH ₃ CL _e ($\mu\text{g}/\text{m}^3$)	Minimum CL (Kg/N/ha/yr)	Maximum CL (Kg/N/ha/yr)
European dry heaths	Yes	Yes	1	5	15
Old sessile oak woods with Ilex and Blechnum in the British Isles	Yes	Yes	1 or 3	10	15
Myotis bechsteini			3	10	15
Barbastella barbastellus			3	10	15
Rhinolophus hipposideros			3	10	15
Lucanus cervus			3	10	15
Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion roburi-petraeae or Ilici-Fagenion)	Yes	Yes	1 or 3	10	15

Spanning a large area of the national park, different areas of the Exmoor Heaths SAC experience different levels of agricultural intensity (Figure 75), with areas in South Exmoor having slightly higher modelled agricultural emissions than the rest of the SAC. Although, as heathlands are largely in more upland or coastal areas, agricultural activity is generally limited by constraints from the terrain or climate, meaning that overall, agricultural emissions are relatively low within the SAC. Hill farm managers in Exmoor have confirmed this, with reports of mainly extensive and low intensity grazing practices on the upland heaths. Outside of the SAC there are

²² [Exmoor Heaths - Special Areas of Conservation](#)

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a few more agriculturally intensive areas according to the NAEI data: near the centre of the National Park near Exford village, and just outside of the National Parks southern boundary near the South Exmoor SSSI. Within a 2km buffer area of the SAC, the dominant agricultural emission sources are modelled to be beef and sheep (Figure 76).

Most of the SAC are modelled to experience concentrations below $1 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$, the critical level for lichens and bryophyte species present on the site (Figure 77). The areas of the SAC towards the south of the National Park are on the border of being above this threshold. Overall, the estimated area-weighted average NH_3 concentration is $0.9 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$, ranging from 0.6 to $1.7 \mu\text{g NH}_3 \text{ m}^{-3} \text{ yr}^{-1}$.

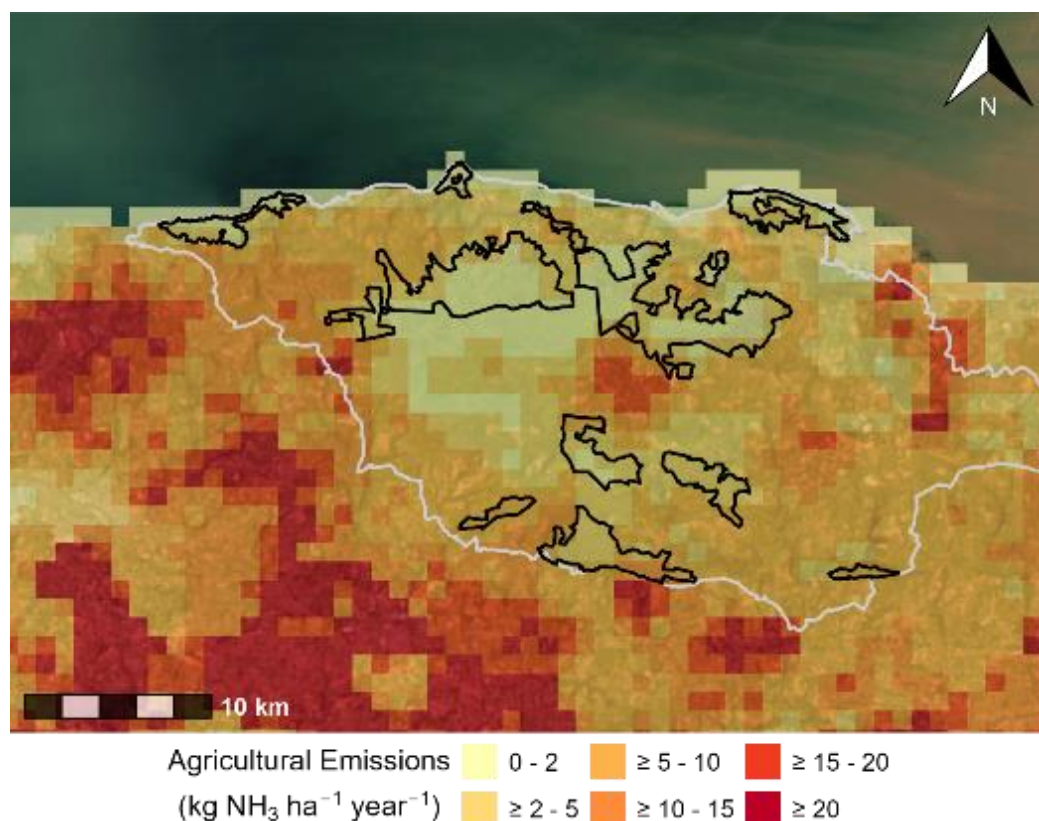


Figure 75: Gridded 2021 agricultural emission estimates for Exmoor Heaths SAC (1km x 1km grid resolution, NAEI)

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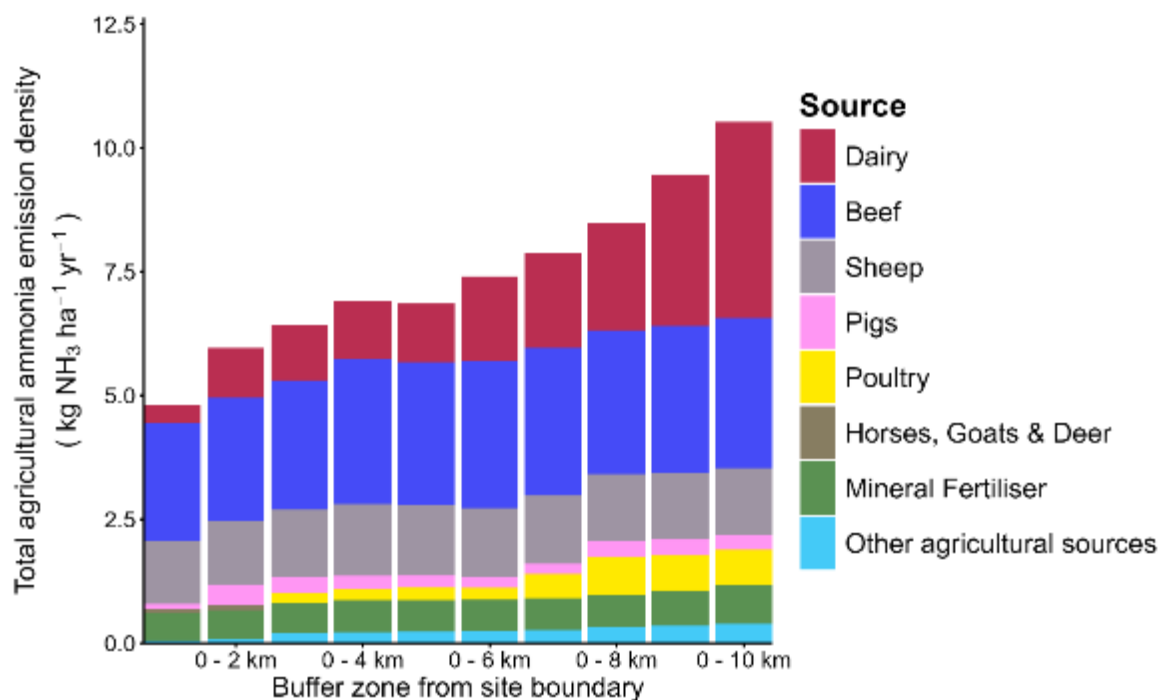


Figure 76: Agricultural emission densities Exmoor Heath SAC. The category ‘other agricultural sources’ refers to any emissions sources which are disclosive (i.e., data points from less than five agricultural holdings).

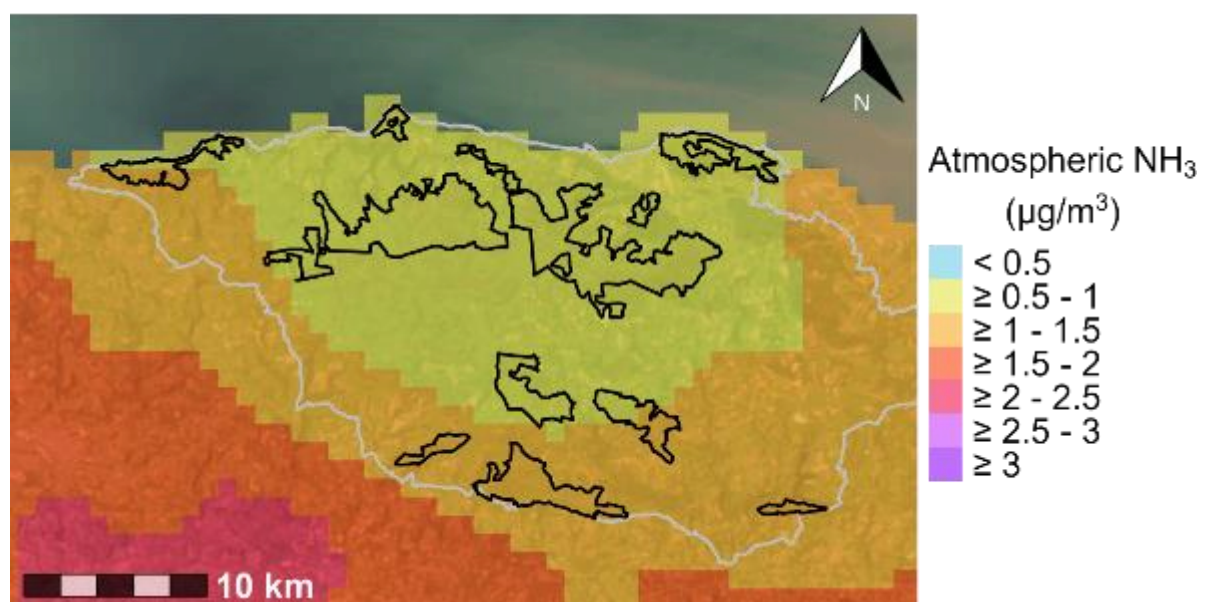


Figure 77: Mean modelled NH_3 concentrations for 2019-2021, calibrated EMEP4UK ammonia concentrations, for Exmoor Heaths SAC

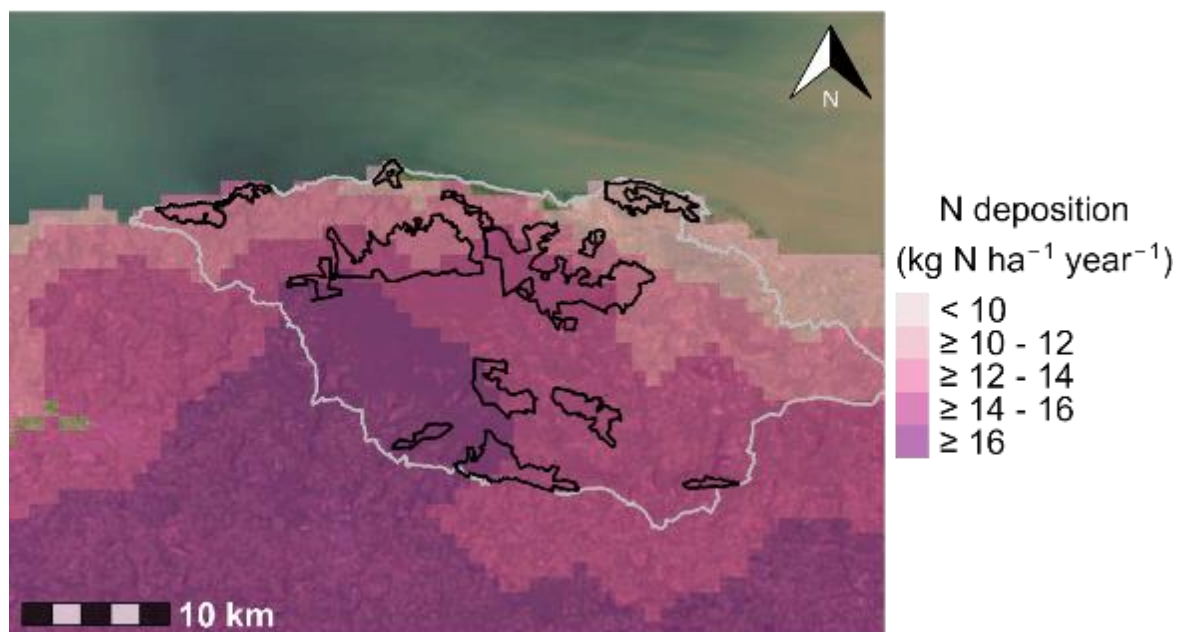


Figure 78: CBED mean modelled N deposition for 2019 – 2021 onto low-growing semi-natural vegetation, for Exmoor Heath SAC

Only nitrogen deposition estimates onto low-growing and semi-natural features like heathlands are described in the following section, as this matches the matches the Annex 1 habitats classified by the JNCC for the Exmoor Heaths SAC. The area-weighted mean rate of N deposition to low-growing vegetation (where present) to the site is estimated at 14.4 kg N ha⁻¹ yr⁻¹, with ~ 46 % of this being in the form of dry deposition (CBED model output averaged for the years 2019 – 2021, Figure 78).

The 2018 source attribution dataset estimates that the largest contribution to nitrogen deposition (to low-growing features) received by Exmoor Heaths SAC is from livestock emission sources in England (~39%, Figure 79). Livestock sources from across the rest of the UK contribute a relatively small proportion. Transport contributes ~6% of the total N deposition received. A substantial proportion of nitrogen deposition is also estimated to originate from sources outside the UK, with European sources contributing ~19%, and international shipping ~10%. These sources would require international intervention to reduce their contribution to N deposition received by the site. Of the reduced nitrogen (NH_x) deposition, which is primarily from agricultural sources, two thirds of dry deposition from more local sources comes from livestock in England. Followed by just under 10% from other agriculture.

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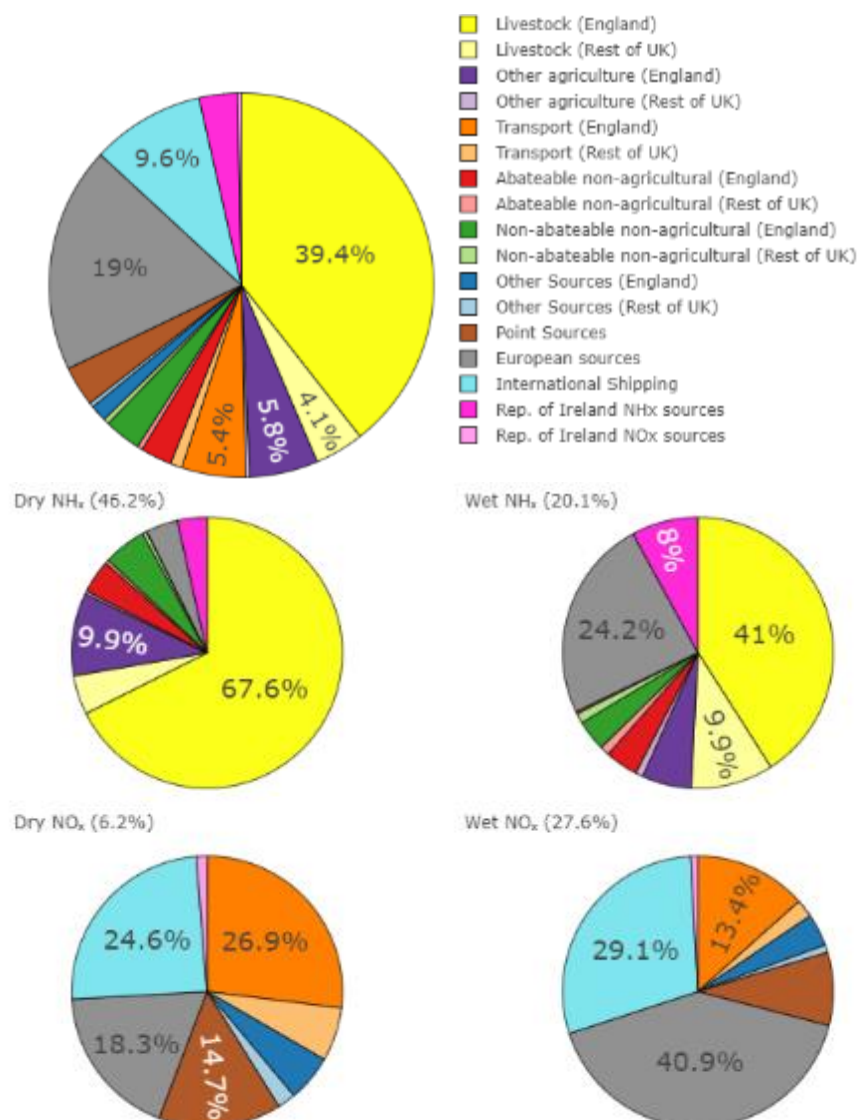


Figure 79: Source Attribution (2018) N deposition received by low-growing semi-natural vegetation for Exmoor Heaths SAC. Top pie chart described total N deposition, which is then broken down into its constituent parts in the four charts below.

5. Mitigation measures

The evidence presented so far shows clearly that emission reduction efforts within the National Park would not be sufficient to bring all designated sites out of exceedance of their relevant critical levels and loads for nutrient N. This is due to the atmospheric N input to the National Park coming from a mix of local, regional, national and international sources. Therefore, a joined-up approach is needed with mitigation actions at several different scales to be considered:

- **local measures** within the National Park, where applicable/possible, recognising that the extensive upland areas are already largely low-emission areas. **This is particularly relevant for any local hot-spot emission sources in close proximity to the designated sites within the National Park** (as outlined for each protected site Section 4 Protected Site Profiles).
- potentially wider **regional measures**, where the areas immediately surrounding the National Park can contribute substantially to a reduction of atmospheric N input, through the application of targeted mitigation measures
- **national-scale measures**, where, for example, the UK has enshrined in law a commitment to reduce NH₃ emissions by 16% by 2030, compared to a 2005 baseline (National Emissions Ceilings Regulations, NECR). The NECR also have targets for NO₂ emission reductions, and there are climate change related efforts under Net Zero ambitions to phase out fossil fuel combustion, thereby also decreasing NO₂ emissions. NO₂ contributes to both elevated local concentrations and dry deposition near combustion sources. There is an overlap here with the wider regional measures, however emission reductions further afield (UK-wide) will also contribute to reduced N input to the National Park.
- **International emission reduction commitments** (e.g. under the National Emission Ceilings Directive, NECD) are expected to decrease atmospheric N input to the National Park from across continental Europe, the Republic of Ireland and international shipping.

5.1 National Scenario modelling

Scenario modelling can quantify the potential impacts of bundles of measures of varying levels of ambition on atmospheric N input to designated sites. The Nitrogen Futures project²³ investigated 15 UK scenarios for 2030 and beyond, from a 2017 baseline (Recent related work underway will not be published during the project and is therefore not available for use in this report). The existing model outputs from the earlier project were re-analysed below to show expected changes to the two SACs in the Exmoor National Park area due to, e.g. the UK meeting its NH₃

²³ <https://jncc.gov.uk/our-work/nitrogen-futures/>



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and NO₂ emission reduction targets under the NECR by 2030 (3 scenarios below described in section 2.1, under N Futures modelling).

Exmoor and Quantock Oakwoods SAC

The latest deposition modelling for the SAC suggests a total N Deposition of 21.2 kg N ha⁻¹ yr⁻¹. Under the 2030 Business As Usual (BAU) scenario, which estimated the impact of currently agreed national policies, **total N deposition reduced by 5%** (equivalent to a reduction of ~1kg N ha⁻¹ yr⁻¹) for the Exmoor and Quantock Oakwoods SAC. Notably this was modelled to be driven by reductions in of all forms of nitrogen deposition except for dry deposition, which is more reflective of local scale emission sources (Figure 80). **Under the NAPCP scenario, a greater reduction of 15%** was achieved for the site which is equivalent to ~3 kg N ha⁻¹ yr⁻¹ down to 18 kg N ha yr.

The current (baseline) mean ammonia concentrations for Exmoor and Quantock Oakwoods SAC are 0.9 µg NH₃ m⁻³. Under the NF BAU scenario, **ammonia concentrations are modelled to increase by 8% to near critical level of 1 µg** (Figure 81), likely due to ammonia concentrations being predominantly controlled by local emission sources which would not be addressed in the national measures in this scenario. The NAPCP scenario which models a further implementation of measures to reach the UK's NECR targets, shows an **ammonia concentration reduction of 6% from baseline**.

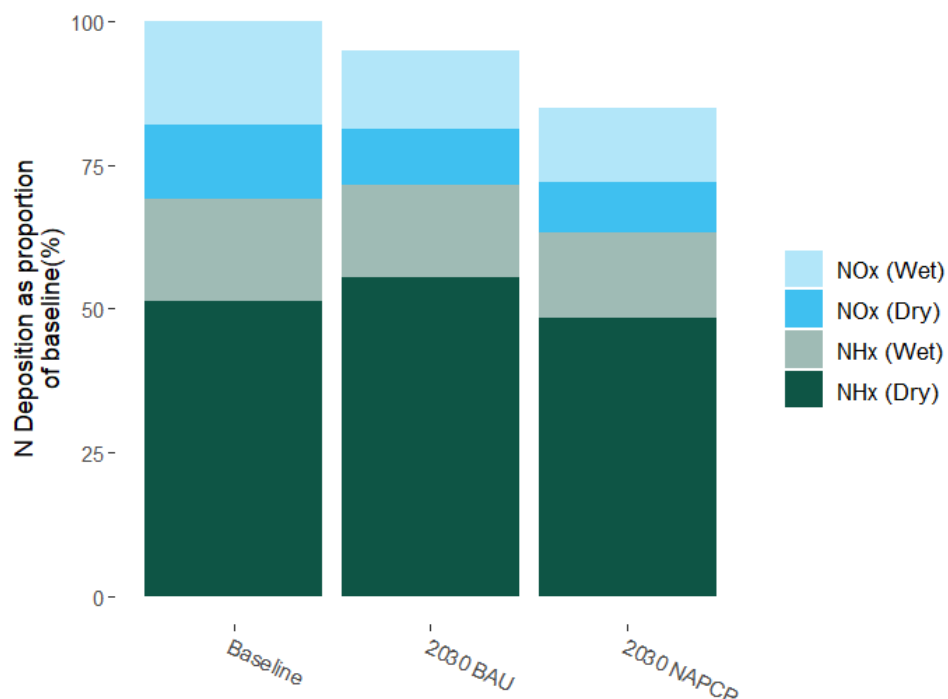


Figure 80: Nitrogen futures 1 scenario modelling changes in N Deposition for Exmoor and Quantock Oakwoods (deposition onto woodland features).



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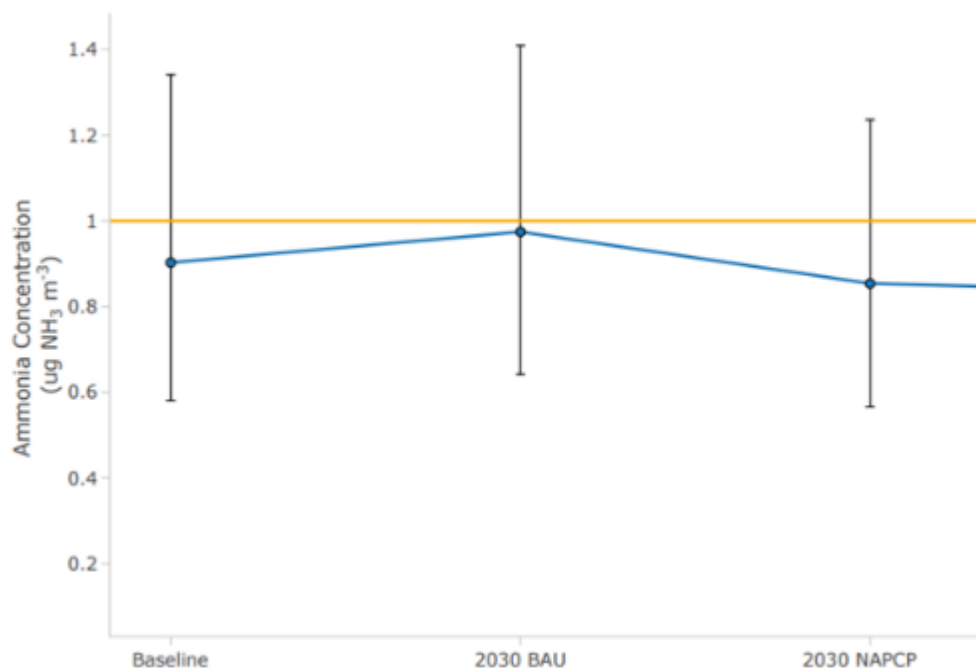


Figure 81: Nitrogen futures 1 scenario modelling changes in ammonia concentration for Exmoor and Quantock Oakwoods, compared to 1 μg critical level (yellow line).

Exmoor Heaths SAC

The latest deposition modelling for the SAC suggests a total N Deposition of $14.4 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. Modelling the impact of currently agreed national policies, the **2030 BAU scenario reduced total N deposition by 6%** (equivalent to a reduction of $\sim 0.9 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) for the Exmoor Heaths (**Figure 82**). Under the NAPCP scenario, **a greater reduction of 15%** was achieved for the site which is equivalent to $\sim 2 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ down to $12.3 \text{ kg N ha}^{-1} \text{ yr}^{-1}$.

The latest EMEP model output suggests a mean ammonia concentration of $0.9 \mu\text{g NH}_3 \text{ m}^{-3}$ for Exmoor Heaths SAC. Under the **NF BAU scenario, ammonia concentrations are modelled to increase by 6% to near critical level of 1 μg** (**Figure 83**), likely due to ammonia concentrations being predominantly controlled by local emission sources which would not be addressed in the national measures in this scenario. The NAPCP scenario which models a further implementation of measures to reach the UK's NECR targets, shows an **ammonia concentration reduction of 6% from baseline, similarly to Exmoor and Quantock Oakwoods SAC**.



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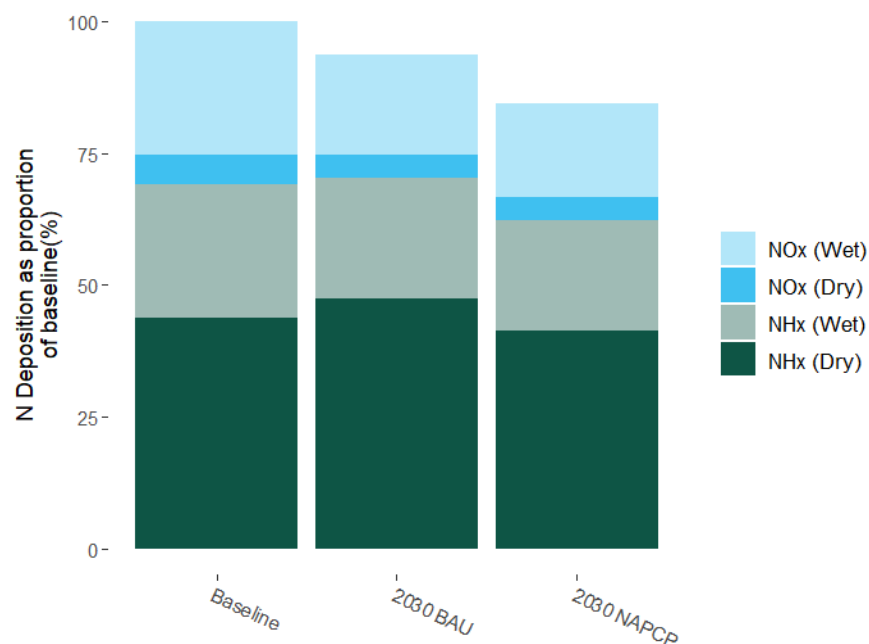


Figure 82: Nitrogen futures 1 scenario modelling changes in N Deposition for Exmoor Heaths SAC (to low-growing semi-natural vegetation)

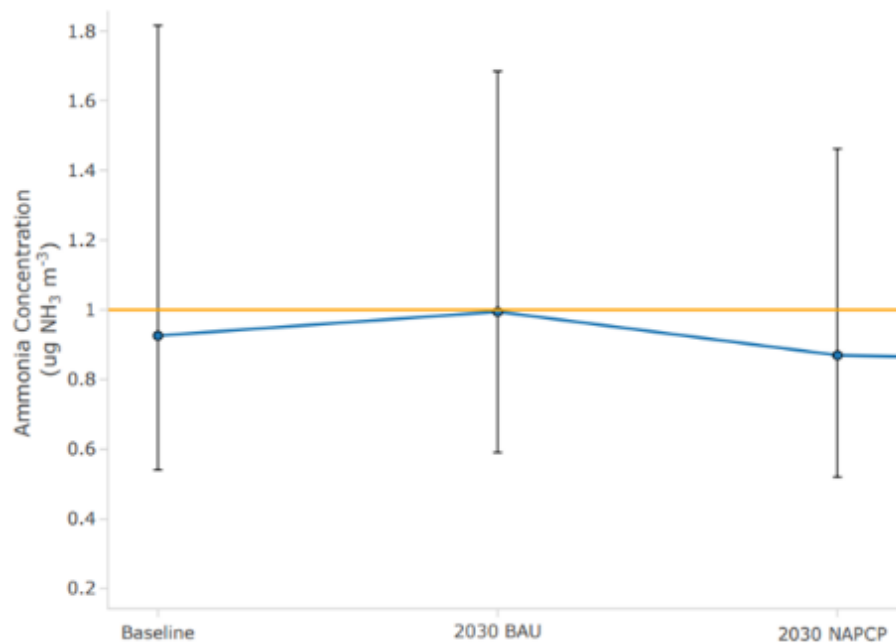


Figure 83 Nitrogen Futures 1 scenario modelling changes in ammonia concentration for Exmoor Heaths SAC, compared to 1 $\mu\text{g NH}_3$ critical level (yellow line).



5.2 Local Measures

As NH_3 concentrations and local dry deposition rates decrease quickly with distance from individual emission sources (due to the high reactivity of ammonia and its tendency to deposit relatively quickly), the largest gains of any mitigation measure can be seen in the immediate surroundings of such sources. This reduces both peak concentrations and longer-term average values. To maximise the efficiency of any mitigation effort within the National Park, **it is therefore recommended that measures are spatially targeted and tailored towards specific emission sources close to N-sensitive protected sites**. The site profiles in Section 4 detail the levels of atmospheric NH_3 each protected site is likely to be experiencing; any areas of higher concentrations and deposition; emission hotspots; and local sources of ammonia, according to the available evidence (from national scale modelling).

The following paragraphs detail the types of practical and deliverable measures which could be implemented within Exmoor National Park. This list needs to be envisaged as a “toolbox” of available measures, where measures can only be implemented if they are suitable for a specific source and where best practice is not already in place.

Implementing these measures requires careful planning and site-specific considerations. The main NH_3 emission sources from agriculture are described below, separately for the different components of the manure management cycle and other activities such as the application of mineral fertiliser. For measures relevant to other N emission source sectors, such as transport, a short summary of options is provided below, too.

Many of the mitigation measures outlined below are consistent with existing agri-environment schemes and broader sustainable land management initiatives. In several cases, elements of low-emission practice are already being implemented within the wider landscape. Building on this existing foundation through targeted advisory support, incentives and collaborative approaches may improve uptake and ensure that actions are proportionate and locally appropriate.

Agricultural measures

Livestock housing and manure storage

Emission reductions can be achieved from housing and storage by regular scraping of surfaces in livestock houses and yards. Covering heaps of FYM to keep manure dry can reduce emissions, either via roofing or covers. Alternatives to traditional open-air manure heaps should be considered based on available resources. According to feedback from land managers in the National Park, FYM systems and the associated storage of manure in heaps are common, particularly at the hill farms. This suggests that changes to manure heap management may be a suitable emission reduction measure, particularly near protected sites on upland farms like North Exmoor SSSI.



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There are limited opportunities for emission reduction from slurry management within the National Park, as most farms use FYM systems. However, for any farms with slurry systems within the National Park, there are a number of emission reduction measures that could be applied. These include covering slurry stores with crusts, floating or rigid covers, or replacement with slurry bags. If/where larger pig and poultry sheds exist, air scrubbers and other emission reduction measures should be considered to minimise emissions, as these are usually very strong local emission sources.

Emission reductions can also be achieved through management of livestock housing. Increasing the frequency of manure removal from cow cubicle housing reduces emissions to air from the exposed excreta.

Table 16: Summary of livestock housing and manure storage measures, adapted from IPENS049 report (2015) and Bittman *et al.* (2014).

Emission Source	Measure	Emission Mitigation Effect
Cattle farms/FYM/poultry manure storage	Farmyard manure heaps are covered with an impermeable sheet for the duration of storage.	60%
Manure heaps	Siting of temporary manure heaps in fields away from the vicinity of designated sites (at least 500m), also taking account of local topography and prevailing winds.	Reduction in NH ₃ concentration and N deposition at the designated site , 0 % reduction in emissions overall as emission source only moved elsewhere
Slurry tanks	Covering above-ground slurry tanks to reduce gaseous emissions to the atmosphere. Different cover types include: - Tight lid - Plastic cover - Floating cover	Tight lid = > 80% Plastic cover = > 60% Floating cover = > 40%
Cattle housing	Increased frequency of removing manure from the floor of dairy cow cubicle housing	15%
Cattle housing	A “grooved floor” system for dairy and beef cattle housing, employing “toothed” scrapers running over a grooved floor,	25 - 46%



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	This results in a clean, low-emission floor surface with good traction for cattle to prevent slipping.	
Cattle housing	Optimal barn climatisation with roof insulation (RI) and/or automatically controlled natural ventilation (ACNV)	20% due to the decreased temperature (especially in summer) and reduced air velocities

Any reductions in atmospheric N emissions at the housing and manure storage stages result in increased N retention in slurry and manure products. While this is beneficial for land managers in terms of cost savings for nitrogenous fertiliser and efficiency of manure usage, it does mean there is increased potential for N losses at the land-spreading stage which should be considered and avoided (by using low-emission spreading equipment). This highlights the need for consideration of the whole agricultural management system when implementing mitigation measures.

Livestock grazing

Grazing emissions are one of the smaller components of NH_3 emissions from livestock and are a much more diffuse source than livestock houses or manure stores. Generally, the upland areas in Exmoor National Park have long-established and relatively extensive grazing practice on low-input grasslands which may be a contributor to the high heathland biodiversity value. For higher input grasslands, decreasing N input to grassland/lower stocking densities grazing close to a protected site would be helpful.

Regular soil testing and baseline assessments, such as soil health evaluations and carbon audits, provide essential evidence to guide grazing management by clarifying soil condition, nutrient balance, and areas for improvement. Enhancing soil structure and increasing organic carbon through practices like diversifying sward composition with flower-rich or deep-rooting species, reducing soil disturbance, and applying appropriate grazing regimes can improve nutrient retention, reduce run-off and leaching risk, and support more resilient grassland systems. These improvements in soil function not only lower the need for inorganic fertiliser inputs, but also enhance nitrogen use efficiency and help reduce ammonia volatilisation and associated emissions. In this way, measures that build soil carbon, strengthen pasture diversity, and promote healthy grazing dynamics can simultaneously support productivity, extend grazing seasons, and deliver climate and air-quality co-benefits while improving the ecological resilience of grassland habitats.

Where grazing occurs in close proximity to protected site boundaries, careful management of stocking density may help reduce local ammonia concentrations. Reducing livestock numbers, adjusting grazing rotations, or avoiding congregation areas immediately adjacent to sensitive habitats could limit direct ammonia exposure. Targeting these adjustments spatially, particularly along the southern boundaries of affected sites where agricultural activity is most intensive, may provide disproportionate benefits relative to broader landscape-scale changes.



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Generally, increasing the period when dairy animals are in the field (especially when 24 hours a day) can reduce NH₃ emissions, since this avoids NH₃ emissions associated with housing, manure storage and land spreading of slurries and/or solid manure. However, the potential for increasing time spent grazing is potentially limited at Exmoor National Park due to the lack of dairy farming and factors such as topography, climatic conditions, economic considerations.

Table 17: Summary of livestock grazing measures, adapted from IPENS049 report (205) and Bittman *et al.* (2014).

N Source	Measure	Emission Mitigation Effect
Grazing	Decreasing stocking densities near protected site boundary	Reduction in NH ₃ concentration and deposition at the site , 0 % reduction in total emissions (as animals will be located elsewhere instead)
Livestock Housing	Increasing the period when animals are in the field compared to housing can be an effective measure to decrease NH ₃ emissions.	Total annual emissions (from housing, storage and spreading) from dairy systems may decrease by up to 50% with nearly all-day grazing, compared with animals that are fully confined.

Landspreading of livestock manures and slurries

Low-emission land spreading techniques can make a considerable difference to emissions from this component of the manure management cycle. For FYM application on arable land, rapid incorporation should be implemented. This is, however, not an option for FYM application to established grassland. For slurry application, low-emission land spreading techniques and equipment such as shallow injection (subject to suitable soil conditions), trailing shoe or trailing hose can substantially reduce emissions, thereby retaining more of the N content in slurry for take up by the vegetation. Before spreading, weather conditions should be checked, as laid out in the Code of Good Agricultural Practice, as ammonia emissions tend to be smaller in cool and wet conditions and after light rain (though water-logging of soils can make spreading conditions unfavourable). Ammonia emissions can therefore be reduced with land-application timing restricted to forecasted periods when lower NH₃ emissions are likely.

Table 18: Summary of measures for landspreading of livestock manures and slurries, adapted from IPENS049 report (2015) and Bittman *et al.* (2014).

N Source	Measure	Emission Mitigation Effect
Arable and grassland	Apply slurry to land via open-slot shallow injection instead of surface	Open-slot injection - 70% Closed-slot injection - 90%



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	broadcast application (subject to suitable soil conditions).	
Arable and grassland	Surface applied slurry is incorporated into the soil shortly after application by either plough, disc or tine (not suitable for established grass swards)	< 4 hrs (Plough - 65%; Disc/tine - 50%) < 24 hrs - 30%
Arable and grassland	Surface applied FYM is directly incorporated into the soil shortly after application by either plough, disc or tine. Emission reduction depends on the time between application to surface and incorporation (not suitable for established grass swards).	Immediate ploughing – 90% (efficacy impacted by degree of manure burying) < 4 hrs (Disc/tine - 45%) < 24 hrs - 30%

Mineral Fertiliser application

For most of the protected sites in Section 4, mineral fertiliser contributes a relatively small proportion of atmospheric N pollution in Exmoor. With the high cost of synthetic fertiliser and the generally extensive grazing management already occurring in Exmoor National Park, usage is relatively limited. However, where mineral fertiliser is being used, low-emission spreading techniques should be implemented. In terms of mineral fertiliser types, replacing urea with ammonium nitrate or using urea fertiliser inhibitors can reduce the proportion of the N applied that volatilises as NH_3 into the atmosphere (instead of being taken up by plants). Checking soil nutrient requirements and applying fertilizers at the right stage of plant growth is fundamental to reducing the amount of excess N available for emission (i.e. nutrient management planning). From feedback from the Exmoor Hill Farming Network, some soil testing is already occurring.

Table 19: Summary of mineral Fertiliser application measures, adapted from IPENS049 report (2015) and Bittman *et al.* (2014).

N Source	Measure	Emission Mitigation Effect
Arable and grassland	Low-emission spreading techniques such as incorporation of fertiliser into the soils	-
Arable and grassland	Replacement of urea with ammonium nitrate	90%
Arable and grassland	Use of urease inhibitors (if using urea-based fertiliser)	70% for solid urea, 40% for liquid urea ammonium nitrate



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Arable and grassland	Checking soil nutrient requirements	Relative to reduction in application of fertiliser
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Secondary measures

Secondary measures include designing tree shelter belts, using species more tolerant of high NH_3 concentrations, to increase dispersion and partly recapture NH_3 near emission sources or designated sites. According to modelling work in JNCC Nitrogen Futures 1, the inclusion of tree shelter belts reduced total reactive N deposition onto SSSIs in England by 20.8%²⁴. This does not reduce the total NH_3 emissions. Another option could be to extensify fields close to designated sites that currently receive high N applications. The Environmental Land Management Schemes contain measures such as payments for low (or no) N input, conversion of agricultural land to semi-natural land. The creation or enhancement of low-emission buffer zones between intensive agricultural land and sensitive habitats can work to reduce ammonia exposure. This could include areas of reduced fertiliser input, lower stocking density, woodland or shelter belt planting, or transition to lower-intensity land uses. Such buffer areas can help intercept, dilute and disperse ammonia before it reaches protected sites, particularly where they are strategically located in relation to prevailing airflow patterns and emission sources.

On-site measures

On-site measures in different priority habitats could also be adopted to mitigate harm from continued N deposition and/or elevated concentrations. Such measures could include low-intensity grazing to keep tall and/or fast growing vegetation in check and maintain species of conservation concern, including insects. Such an approach was inadvertently demonstrated in Californian grasslands (Weiss, 1999), but the concept is likely transferable to grassland systems in Exmoor. In the Californian grasslands, Weiss (1999) observed that areas subject to moderate grazing pressure helped limit the dominance of invasive, fast-growing grasses that thrived under increased N input. These invasive species shaded out native plants and reduced habitat suitability for endangered insects that relied on them. Although the Californian system differs ecologically and climatically from Exmoor, the underlying principle is transferable: carefully managed grazing can help counteract the competitive advantage gained by fast-growing species under elevated nitrogen, preserve plant community structure, and maintain biodiversity. However, the amount of grazing that maintains the priority targets for a given habitat will depend upon the level of N deposition, the specific targets and the current state of the habitat, as well as potentially legacy conditions. The complexity of this sensitivity to levels of grazing has also been demonstrated in *Calluna vulgaris* (heather) moorlands, where the response of vegetation to grazing pressure is closely linked to nutrient status. Research has shown that light to moderate grazing can help

²⁴ <https://data.jncc.gov.uk/data/04f4896c-7391-47c3-ba02-8278925a99c5/JNCC-Report-665-FINAL-WEB.pdf>



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maintain the dominance of heather by reducing the competitive pressure from nitrophilous grasses, particularly in areas with elevated nitrogen deposition. This creates a buffering effect, allowing the system to tolerate slightly higher nitrogen loads without undergoing rapid or irreversible degradation (Alonso et al., 2001; Hartley et al., 2003; Hartley and Mitchell, 2005). Although this effect is subject to site-specific conditions and factors such as soil moisture and organic matter content. Overall, appropriate grazing regimes can function as a mitigating mechanism, offsetting some of the damaging effects of nitrogen enrichment by maintaining ecosystem structure and preventing grass encroachment. However, the sensitivity of these systems to both over- and under-grazing highlights the importance of site-specific management planning, supported by monitoring and adaptive responses to ecological feedback. The goal is to strike a balance: applying enough grazing pressure to suppress undesirable growth while avoiding damage to keystone species and soil integrity. Grazing management should therefore aim to maintain appropriate vegetation structure and prevent dominance by nitrogen-tolerant species, ensuring that mitigation measures do not unintentionally exacerbate ecological change.

These and other on-site management measures were recently assessed in Britton et al. (2024), who highlighted the limited number of studies on how to accelerate system recovery from N deposition. Their evidence review found studies for dunes, fens, grasslands, heaths (alpine moss and dwarf-shrub) and forests. These studies were mainly from the south of the UK or Europe and considered few ecosystem responses and/or recovery indicators (see Perring et al. 2024), and only a small number of options. Their conclusions included that turf-cutting and liming could reduce acidification in grasslands and forests (although this may not be appropriate for Exmoor with currently management practices), while grazing or mowing could be used to improve sward structure and maintain species richness in grasslands. Turf cutting involves the removal of surface vegetation and topsoil to eliminate accumulated nitrogen and re-establish early successional conditions on nutrient-poor substrates. It has been shown to be an effective restoration method in nitrogen-saturated heathlands and grasslands in the Netherlands, where reactive nitrogen pollution is particularly high and critical loads are frequently exceeded (van den Berg et al., 2003). This can have a combination of positive and negative impacts, favouring species with a persistent seed-bank at depth, which in heathlands may favour *Calluna* over invasive graminoids such as *Molinia*, although it could also lead to the removal of seeds of species with high conservation value (e.g. van den Berg et al., 2003; Dorland et al., 2005). Liming is sometimes used alongside cutting to raise soil pH and counteract acidification caused by atmospheric deposition of nitrogen and sulphur. Experimental work has shown that combining liming with turf cutting can improve conditions for germination and establishment of threatened plant species in both dry and wet heathlands (Dorland et al., 2005). However, lime may also increase soil organic matter decomposition rates and lead to the release of more available N, worsening eutrophication and leading to the invasion of ruderal species. Liming should only be considered where acidification is clearly an issue, i.e. where pH has been measured and found to be



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below a critical value, for instance 4.4 pH units which is considered the critical value for peaty soils (Hall et al 2015).

Burning, grazing and mowing were all shown as potential options to reduce N stocks in heathland. Burning reduces biomass and removes above-ground nitrogen, but not soil N. In an upland heath in north Wales, UK, Pilkington et al. (2007) found that burning removed a large proportion of accumulated N from the vegetation and litter but did not lead to any short-term decrease in N stores in the soil. Pilkington et al. (2007) suggest that burning approximately every 10 years can help remove nitrogen that has built up in the vegetation and litter, particularly in areas with moderate nitrogen pollution (up to 56 kg N per hectare per year). This can be a useful tool for managing nitrogen levels in the short term. The study also warns that widespread or uncontrolled burning, especially in areas with very high nitrogen pollution, could make the situation worse. If the soil is already saturated with nitrogen, burning can cause more of it to wash down through the soil and into nearby water. This can lead to problems like acidification, nutrient overload (eutrophication), and discoloured or “brown” water in streams and rivers. These findings show that while burning can be useful in some situations, it needs to be used carefully and in the right places.

Of concern was the broader finding that adopting some of these measures could have inadvertent side-effects, such as loss of soil carbon, reduction in habitat suitability for fauna, and nitrate leaching following disturbance, with the potential that ‘trying to solve one problem could create another’ (Britton et al., 2024). This suggests that **Exmoor NP may need to encourage trials of different options to inform site-specific management and conduct appropriate benefit-cost analyses prior to implementing measures widely.**

Non-agricultural measures

While agricultural NH_3 is the largest form of local N emissions (Figure 3), there are some areas with non-agricultural sources where measures could be considered (particularly for sources of NO_2). The most effective measures targeting deposition from road transport are likely to be from national level measures such as the promotion of greener technologies (alternative fuels and end of pipe technologies) or personal transport choices. High-cost measures such as the alignment of transport links do not seem to be proportional to the risk, given the relatively low emissions from the A39 and A396, compared with other major UK roads. In summary, targeting agricultural emissions is thought to be most cost-effective way to decrease atmospheric N input to the designated sites through from N deposition. However, local NO_2 concentration gradients would be decreased by reduced traffic on busy roads in the area. It is expected that increased adoption of non-fossil fuel transport (such electric vehicles) under Net Zero efforts will result in decreased NO_2 concentrations over time.



5.3 Monitoring

Monitoring atmospheric nitrogen is a key part of an effective mitigation strategy as it allows a better understanding of ecosystem impacts, supports the identification suitable locations for spatially targeted abatement measures and allows for the quantification of the impact of any implemented measures. An effective monitoring strategy would aim to include at least 12 months of monitoring, before any mitigation measures are implemented, to establish a baseline and to be able to detect change. Typically, atmospheric nitrogen is monitored by sampling NH_3 concentrations, NO_2 concentrations and wet deposition. A monitoring framework could also be expanded beyond atmospheric N, to ecosystem measurements. Recovery of habitats and species in response to decreased pollution may be slow, given legacy N accumulation, although epiphytic lichens and mosses can respond more rapidly.

NH_3 concentrations

Over 85% of UK NH_3 emissions originate from agricultural sources, therefore most local emission sources are expected to be from farming activities. Compared to other forms of atmospheric N, NH_3 concentrations are highly variable over both space and time, with concentrations decreasing along a gradient away from any sources. Given there are a large number of potential sources of NH_3 across the wider Exmoor landscape, a relatively high density of sampling sites should be established if possible. Protected site NH_3 monitoring is generally undertaken with passive diffusive samplers to enable cost-effective spatial coverage, as monthly cumulative measurements. With continued monitoring, NH_3 concentrations at protected sites can be compared against relevant ecosystem critical levels.

The approach for determining the location of samplers includes consideration of the spatial variability of emission sources, meteorological conditions and terrain, the location of N-sensitive features and input from local stakeholders on accessibility and land ownership for permission to set up samplers. In general, NH_3 monitoring strategies look to characterise NH_3 concentration gradients within a protected site and the concentrations that sensitive ecosystems of particular interest are exposed to (e.g. locations of rare lichens, temperate rainforest, upland heaths). Exmoor National Park also poses an opportunity to measure the potential influence gamebird releases on NH_3 concentrations (outlined in the Gamebirds section 3.4 of this report).

Specific monitoring locations could be based on the model output datasets for each designated site outlined in the site profile section of this report, alongside expert local knowledge of the area. In addition, satellite imagery was analysed to pinpoint potential point emission sources that would not be obvious from the spatially averaged 1 km by 1 km model output data (examples of this method shown in Case Study section). Sampling strategies for each designated site should consider aspects such as:

- Strong atmospheric concentration gradients away from emission sources



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- Whether there would be enough samplers to capture gradients in concentration profiles
- Locations of high value designated features
- Specific locations of interest (such as near emission sources)
- Accessibility for sampler installation and monthly exchange
- Prevailing winds

NO₂ concentrations

NO₂ is mainly emitted from combustion processes, such as transport, heating, power generation and industrial processes, but also wildfires and controlled burning. NO₂ sampling should be focused along gradients extending away from the busiest/high-traffic roads where they intersect with designated sites, such as North Exmoor SSSI where it intersects with A39 or other tourist hotspot areas/protected sites identified as having potential to be affected by combustion sources.

Wet Deposition

Many ecosystem impacts of air pollution relate to the total load of atmospheric received (from both oxidised and reduced N components), that is total N deposition. Measuring wet deposition of N (in precipitation) requires regular collection and analysis of rainwater. The national wet deposition monitoring network (UK Eutrophying and Acidifying Atmospheric Pollutant (UKEAP) Precipitation Network (Precip-net) (Stephens et al., 2021)) uses an approach of fortnightly sampling, to balance cost effectiveness and accuracy of measurements. Wet deposition of N is less spatially variable than NH₃ or NO₂ concentrations, therefore fewer samplers would be needed to characterise wet N deposition onto Exmoor National Park.

5.4 Mitigation prioritisation method

The following section outlines a methodology to identify emission sources which could be prioritised for assessment of mitigation potential. Examples of this method for three case studies are detailed in Section e), and show how this approach could be used to identify potential farms and areas for spatially targeted mitigation measures. This approach is based on the methodology of site categorisation for atmospheric N measures outlined in the evidence supporting the Improvement Programme for England's Natura 2000 Sites published by Natural England (IPENS049, 2015).

When investigating mitigation potential for a protected site, the current exposure levels to atmospheric N pollution must be considered first alongside the sensitivity of the habitat to each form of N (i.e. assess the critical loads and levels of



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atmospheric NH_3 concentrations and N deposition). The latest modelled NH_3 concentration and N deposition estimates are outlined for each SSSI and SAC in **Table 4** of this report, alongside the most sensitive critical load and level.

Next, the main threats from atmospheric N should be identified for each protected site. The datasets outlined in site profile section of this report were used to understand the drivers of atmospheric nitrogen pollution to each site, as well as whether each source type (agriculture, point sources, roads, etc) allocated was due to substantial sources, or largely due to long-range sources (such as from mainland Europe or international shipping). The IPENS049 report (2015) suggested road transport is considered a significant source of N deposition where vehicular emissions contributed to > 10 % of the total N deposition, in combination with the presence of a major/busy roads (e.g. A-roads) within 200m of a site boundary. This understanding of the main N threats to each site allows for the filtering of measures to exclude those not relevant to the site (e.g. pig farming specific measures, when the predominant agricultural activity is cattle farming in an area).

For the two SACs, Nitrogen Futures modelling output was used to estimate the relative impact larger scale policy implications could have on the sites (outlined in section 5.1). This allowed an initial assessment as to whether local mitigation measures were likely to be worth considering for targeted reduction of atmospheric N at a site, or whether a wider regional or national/international effort would be necessary as the main route for improvements.

While a reliable identification of the main threats and local vs. regional issues can be achieved with the approach described above the detailed selection of potential local measures requires local collaborations and sharing of information on current management systems and practices, prior implementation of low-N systems and measures, etc.

Aerial imagery can be used to identify types and locations of farm buildings which could be prioritised for assessment of mitigation potential. Aerial imagery analysis should be used in conjunction with local knowledge of building uses. For the following case studies, Google Earth imagery was used in conjunction with national datasets summarised in the site profile to identify any additional sources within a 2km boundary of the protected site and provide further information on already identified sources, including distance of sources from the site boundary, visual assessment of local conditions etc. This step could be improved by the incorporation of local knowledge and collaboration with local stakeholders and site managers as to what and where local sources are. This can be particularly relevant when considering that changes may have occurred since an aerial image was taken/dataset was compiled and that manure heaps are common practice with Exmoor and are less likely to be identified using satellite imagery due to their temporary nature.



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Local knowledge is also needed to understand if farmers are already implementing mitigation measures. This needs to be taken into account both for crediting such reductions in local emissions to the farmers and for understanding the relative importance of local emission sources and the potential for further complementary measures. While a more general profile of management practices was formed from the collaboration with Exmoor Hill Farming Network and FWAGSW, the specific spatial targeting of measures requires understanding of management practices on a farm-by-farm basis.

Overall, the following key steps are recommended:

- a) Analysis of atmospheric N deposition (including source attribution, contributions from wet/dry deposition) and NH₃ and NO_x concentrations
- b) Analysis of relevant data for all relevant source attribution for each protected site (Protected Site Profiles).
- c) Familiarisation with the site, aerial images and output from the previous analysis steps (Case Studies), pulling together of site profile and draft list of potentially suitable measures for local targeting (Agricultural measures);
- d) Communication with local site managers and other interested stakeholders to check site profiles and preparation of a revised list of suitable locally applicable measures.
- e) Where data is available, an understanding of local wind conditions can help to understand which potential emission sources are downwind of the protected site.

5.5 Case Studies

The following case studies briefly summarise Site profiles for each site and give examples of aerial imagery analysis and consideration of relevant emission sources through the method outlined Section 5.4. The individual case study sites were selected based on the likely presence of local emission sources (as identified in Site Profiles) and feedback from the ENPA project team.

Case Study 1: North Exmoor SSSI

According to the Site Profile for North Exmoor SSSI, livestock in England was the biggest source of N deposition to the site which could be (partly) influenced by local measures. International shipping and emissions from European sources would be the target of national and international mitigation efforts. In terms of local agriculture, the main emission sources were sheep (representative of extensive upland farming on the heaths) and beef.

The aerial imagery analysis (Figure 84) showed emission sources within 2km of the site boundary were likely to be mainly farm buildings, with 2 potential slurry stores. The density of farm buildings was loosely reflective of the agricultural emission density estimates (Figure 34), with a higher density of farm buildings in the centre of the national park near Exford to the south of the heathlands.



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In terms of road emissions, the source attribution modelling indicated that transport contributed to a relatively small proportion of total N deposition for the wider area. However, some areas of the SSSI are within 200m of the A39.

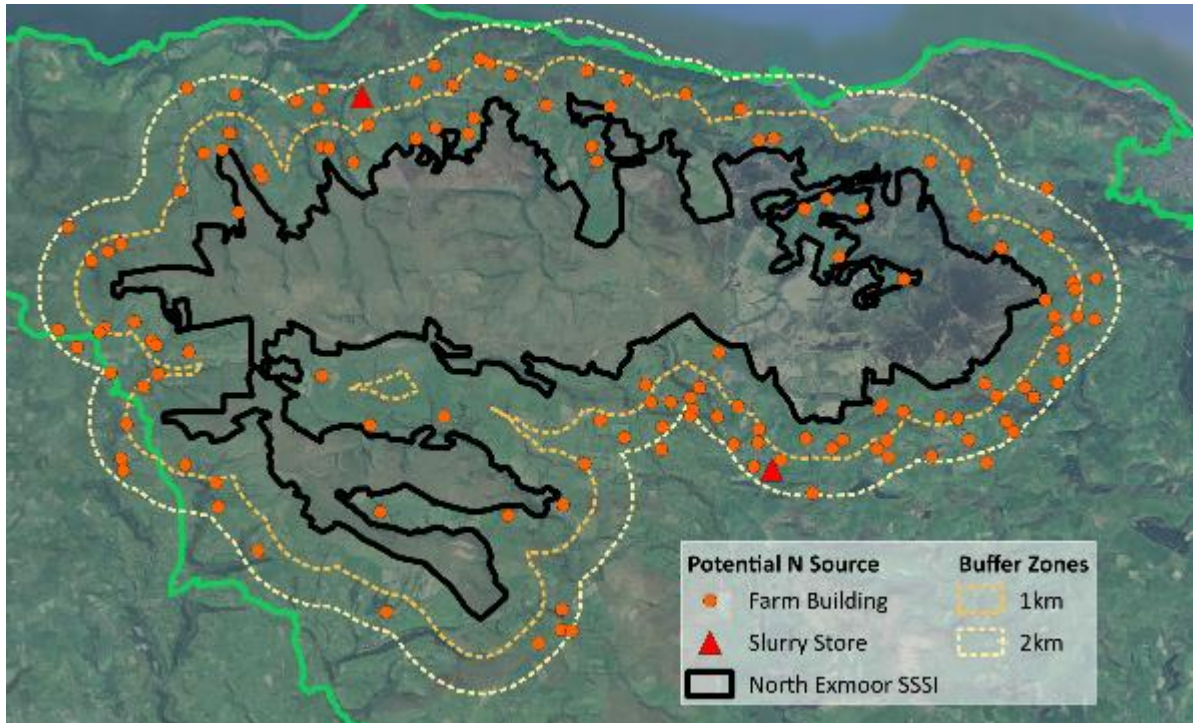


Figure 84: North Exmoor SSSI with potential N sources identified from Satellite imagery, during the desk-based study carried out Dec 2024. Satellite imagery displayed is Bing Aerial.

Case Study 2: Barle Valley SSSI

According to the Site Profile for Barle Valley SSSI (Section 4.1), livestock in England was the biggest source of N deposition to the site which could be influenced by local measures. In terms of local agriculture, the main emission sources were sheep and beef, with the density of beef emissions increasing significantly between 1 and 2 km from the site. There is also potential for a smaller amount of dairy farming within 1-2 km of the protected site boundary.

The satellite assessment identified one potential circular slurry store within 1 km of the site boundary and a number of farm buildings likely used for livestock housing (Figure 85). Several of the potential farm buildings were in very close proximity (<300m) to the SSSI boundary.

In terms of road emissions, the source attribution modelling indicated that transport contributed to a relatively small proportion of total N deposition and the closest A-road is almost 2 km away from the site. Therefore, it is unlikely that road emissions are a significant driver of N pollution across Barle Valley SSSI.



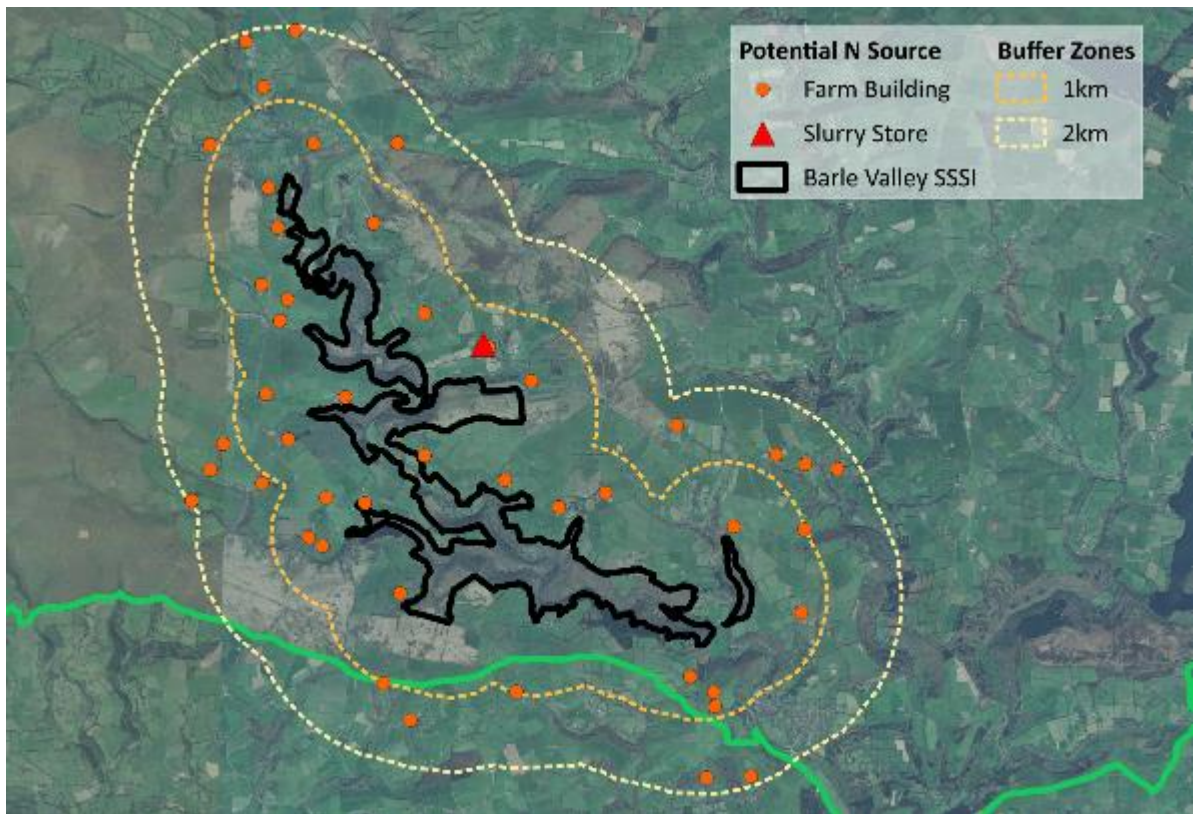


Figure 85: Barle Valley SSSI with potential N sources identified from Satellite imagery, during the desk-based study carried out Dec 2024. Satellite imagery displayed is Bing Aerial.

Case Study 3: Dunster Park and Heathlands SSSI

According to the Site Profile for Dunster Park and Heathlands (Section 4.3), livestock in England was the biggest source of N deposition to the site which could be influenced by local measures. In terms of local agriculture, there are fewer than 5 farms for some emission source categories within 2 km of the site boundary so specific sources could not be identified, however notable pig and dairy emission sources appear within the 0 – 3 km buffer zone from the site boundary.

The satellite assessment identified 4 potential slurry stores within 2 km of the protected site boundary, 3 of which were just outside of the National Park. There were also several farm buildings, likely livestock housing, and the town of Minehead which is modelled to have relatively high levels of non-agricultural NO₂ emissions according to the UK NAEI (Figure 3).

In terms of road emissions, the source attribution modelling indicated that transport contributed to a relatively small proportion of total N deposition. However, some areas of the SSSI are within 200m of the A39.

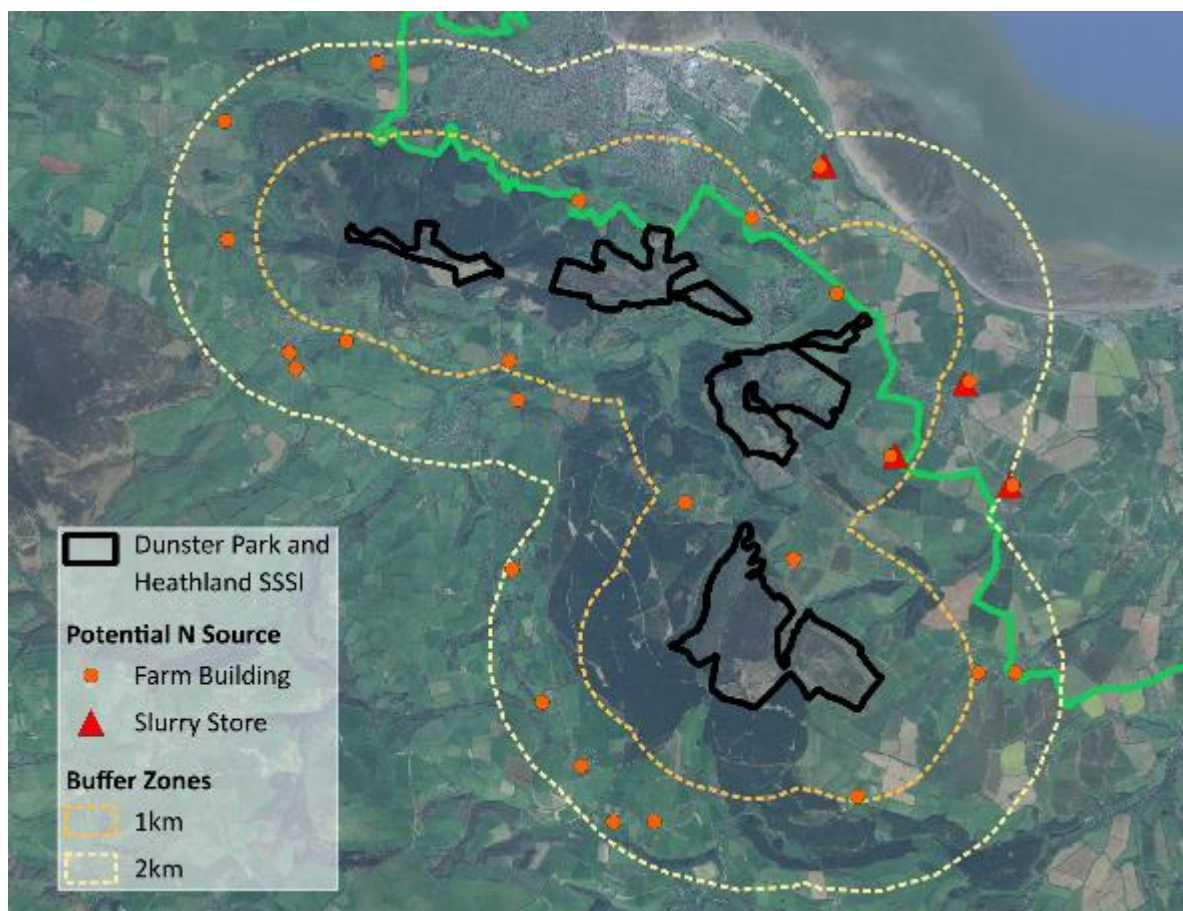


Figure 86: Dunster Park and Heathland SSSI with potential N sources identified from Satellite imagery, during the desk-based study carried out Dec 2024. Satellite imagery displayed is Bing Aerial

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