UK Centre for Ecology & Hydrology

Site Nitrogen Action Plans (SNAPs) for Native Woodland sites in Wales

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Non-technical summary

Air pollution by ammonia and other nitrogen (N) compounds causes damage to habitats and wildlife, especially to sensitive lower plant communities such as lichens and mosses. The SNAPs project investigated the sources of N pollution at five woodland sites in Wales, and assessed how pollution levels and their effects on the woodland could be mitigated. The main source of ammonia at the sites studied was low-intensity beef and sheep farming. While some ammonia disperses over large distances, large amounts of ammonia from a given source are deposited within a few hundred metres, so nearby sources are important. Action is needed to limit regional and international flows of air pollution, but considerable improvements could be made by reducing the emissions from nearby sources of ammonia. The most useful measures are covering slurry stores; hard standing at feeding stations; and injecting slurry into the soil rather than broadcasting onto the ground surface. Some conservation sites are very large, and if particularly sensitive areas can be identified within sites, this could help identify off-site mitigation measures, which can be applied to sources nearest the sensitive area. For woodlands, on-site measures have only limited potential for reducing the ecological impacts of nitrogen pollution. Nitrogen generally increases plant growth and litter production, causing more shading of the ground surface. These effects can be reduced in woodlands by increasing scrub clearance, or grazing. However, these measures may have other effects on the site, and remove very little nitrogen from the system, meaning that it continues to accumulate in the vegetation and soil. Ultimately, nitrogen-sensitive features within woodland sites can only be safeguarded if nitrogen emissions decrease both locally and regionally.

Executive Summary

- The Site Nitrogen Action Plans (SNAPs) for Native Woodland sites in Wales project aimed to identify local sources of atmospheric nitrogen (N) at five selected Special Areas of Conservation (SACs) with woodland features (in particular 91A0 old sessile oak woods with Ilex and Blechnum and 9180 tilio-acerion forests of slopes, screes and ravines) that are sensitive to N. The SACs selected are also the focal area of the Celtic Rainforests Wales Project
- SNAPs were created for each of the selected sites using a draft framework (developed by the project team under several previous projects). The framework uses national datasets to help identify N threats to sites and suggest potential off-site mitigation measures. For all five sites, the amount of N deposition received was estimated to be in exceedance of critical loads for the sites' designated woodland features.
- Modelled ammonia (NH_3) concentrations in some areas of the sites were estimated to be in exceedance of the critical level for lichens, mosses and bryophytes. Overall, ammonia concentrations were less elevated than in other parts of the UK and the implementation of SNAPs may help avoid further deterioration. Some of the sites covered large areas of Wales, and it was therefore difficult to characterise specific N threats, as emission activities (especially agricultural) can be highly spatially variable. Therefore, sub-sites were selected to target areas of interest; these were lichen-rich areas and areas of (relatively) intensive agriculture.
- Most of the sites are situated in areas of extensive agriculture and are characterised by a high proportion of emissions associated with beef cattle and sheep.
- Information on the distribution of sensitive features was only available for one of the five selected SACs (Meirionnydd Oakwoods and Bat Sites SAC). Having this information available for more sites would improve the protection of biodiversity, increase the cost-effectiveness of measures and enable a more confident assessment of risk.
- Potential off-site measures were identified at some of the sites using aerial and satellite imagery. However, local information on current management systems and practices is required prior to identification/implementation of any low-N systems and measures.
- The potential for on-site measures to reduce the ecological impacts of N was assessed through a review of the wider literature, including some data and surveys conducted on Welsh woodlands.
- Unlike in other habitats, the options for on-site management to reduce some of the adverse ecological effects in woodlands is rather limited. Felling and removal of timber removes N from the system, but is generally not a part of the management regime of these Welsh woodland sites for conservation. Scrub clearance can remove some N, but the most realistic management option is grazing or browsing. This can have a mix of positive and negative benefits for the ground flora, but has no impact on epiphytic species.
- A better understanding of the current ecological status of each site may allow a more informed management response. This includes: measurement of ammonia concentrations around and within sites; measurement of soil pH and

soil chemistry including litter C:N ratios; measurement of plant foliar chemistry of trees, ground flora and epiphytes (%N, %P, %C). These ecological measurements can indicate how close to nitrogen saturation each woodland is, which can help to better target local ammonia sources, and the intensity of any on-site measures which can be implemented.

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1 Background

Atmospheric nitrogen (N) deposition and elevated ammonia (NH_3) concentrations are a significant threats to sensitive habitats and species in Wales. Eutrophication and acidification lead to declines in many key species of high conservation value in favour of a smaller number of fast-growing species that can exploit conditions of increased nitrogen supply (e.g., Dise et al. 2011). Many designated sites in Wales, including Natura 2000 sites, remain under substantial exceedance of internationally agreed critical loads for nutrient-N and acidity, and have NH_3 concentrations above their relevant critical levels.

The aim of this study was to produce Site Nitrogen Action Plans (SNAPs) for five Special Areas of Conservation (SACs) in Wales, for identifying local sources of atmospheric N and developing a plan to improve conservation status in respect to N impacts. All five SACs are designated due to having habitats 91A0 (Old sessile oak woods with *Ilex* and *Blechnum*) and / or 9180 (*Tilio-acerion* forests of slopes, screes and ravines) as primary features. The SNAPs were created using a draft framework developed following previous studies including: the Defra RAPIDS project (Dragosits et al. 2014a), the Natural England-funded IPENS-049/-050 projects (Dragosits et al. 2014b, Misselbrook et al. 2014), the NRW-funded AAANIS project (Carnell and Dragosits, 2015), the NIEA-funded EMIND project (Carnell and Dragosits, 2017) and the Defra-funded Nitrogen Futures project (Dragosits et al. 2020). The draft framework helps to identify key atmospheric N threats for protected sites and includes a draft process for identifying the most appropriate mitigation measures and potential delivery mechanisms to implement these.

The framework used for identifying the main sources of atmospheric N received by sensitive habitats is the first step in targeting local mitigation options, given the high spatial variability of NH_3 concentrations and dry N deposition in particular. These options include both source-oriented technical measures (e.g. covering slurry stores, catalytic converters for petrol engines) and landscape oriented measures (e.g. adapting local agricultural practice with low-emission buffer zones around sites, tree belts for dispersion and recapture of emissions, etc.). A wide range of potential measures exists to reduce emissions from agricultural sources. However, in contrast to other countries such as the Netherlands and Denmark, where wide-ranging legislation has been implemented to reduce emissions, there has only been relatively limited uptake of NH_3 mitigation measures in the UK. In addition to off-site mitigation measures (tackling emissions at source or local/landscape scale extensification), restoration measures within the designated sites may also help to reduce either the impacts of atmospheric N deposition by altering habitat suitability for target species or by removing accumulated (i.e. legacy) N itself from the system. These on-site measures were reviewed by Stevens et al. (2013) for all major UK habitat types, including woodlands. On-site measures that can improve habitat suitability include aspects such as manipulating sward height to increase light availability to low-growing species, amending soil pH, and increasing germination opportunities. However, there are fewer options available for woodland habitats and appropriate management may differ for woodland compared with other habitats. Similarly, measures to remove accumulated N from the site differ for woodland compared with other habitat types, and are also dependent on site topography and access. Restoration measures and estimated timescales for recovery will need to take account of the different habitat components within woodland ecosystems (e.g. epiphytes, ground flora, soils, trees) as each will have different responses to both increases and decreases in N deposition (Rowe et al. 2013).

The aim of this project was to develop draft Site Nitrogen Action Plans (SNAPs) for five SACs, which form the focal areas the Celtic Rainforests Wales Project (Table 1).

Table 1: *Natura 2000 sites selected for study. All sites include habitat types 9180 (Tilio-Acerion) and/or 91A0 (Quercus petraea) woodland*

Site Code	Site Name	Site Area (ha)	H91A0	H9180
UK0014789	Meirionydd Oak Woods and Bat Sites	2,813	✓	✓
UK0030117	Coed Cwm Einion	21	-	✓
UK0030128	Cwm Doethie - Mynydd Mallaen	4,122	✓	-
UK0030145	Coetiroedd Cwm Elan/ Elan Valley Woodlands	439	✓	✓
UK0012946	Southern part of Eryri/Snowdonia	3,615 (total area 19,733)	✓	-

The selected Special Areas of Conservation (SACs) contain habitat types H9180 (*Tilio-Acerion* forests of slopes, screes and ravines) and / or H91A0 (Old sessile oak woods with *Ilex* and *Blechnum* in the British Isles) as primary reasons for designating the sites, which were the focus of the project.

The study comprised of three phases:

Phase 1 provided an initial identification of key atmospheric nitrogen threats to sites and their features. This included:

- Identification of the location and sensitivity of habitats, where known, and estimation of woodland area (using land cover maps) where this information was not available.
- Assessment of current and projected rates of atmospheric N deposition and NH_3 concentrations at each site and in the surrounding area. Identification of the main sources contributing to N deposition at each site from the latest source attribution dataset (for the year 2018; unpublished UKCEH data, with funding from the UK government agencies i.e. SEPA, SNH, JNCC, EA, NE, NRW, NIEA).
- Estimating current and likely future agricultural emission densities and identifying agricultural sectors that are likely to contribute to emissions locally.

Phase 2 provided an identification of suitable measures to bring the sites into good condition with regard to atmospheric nitrogen. Including:

- Identifying suitable off-site mitigation measures to reduce atmospheric N input to each site
- Identify suitable habitat management/restoration measures that may help to mitigate remaining (and legacy) nitrogen impacts

Phase 3 developed guidance on implementing SNAPs and an approach for auditing effects of measures on atmospheric N input to sites and their recovery

- Summary assessment of current and likely future threats to the site from atmospheric N input (from Phase 1).
- Recommendations for on-and off-site measures to mitigate effects of atmospheric N on sensitive habitats and species (from Phase 2).
- Suggestions for potential approaches and mechanisms for implementing SNAPs, including the processes required for an ecological audit, to be carried out following implementation of SNAP measures. The audit would evaluate whether the expected reduction in atmospheric N input has been achieved, and whether, together with the on-site habitat measures, has led to the achievement of the conservation objectives for the site.

2 Methodology

2.1 Phase 1: Identification of key nitrogen threats to sites and their features

2.1.1 Location and sensitivity of habitats within sites.

Identifying the location of sensitive features within sites enables improved assessments of critical loads and levels exceedance, especially for larger sites that span a considerable geographic area. This information will also help improve any spatial targeting of measures to protect areas close to sensitive source receptors (i.e. areas of deciduous woodland thought to contain H9180 and/or H91A0).

Detailed habitat maps showing where the designated features are located within sites were only available for Meirionnydd Oakwoods and Bat Sites SAC. This feature map was used for identifying where the woodland features of interest (H91A0, H9180) are situated within the site relative to atmospheric N inputs, which can be highly spatially variable, especially for larger sites (which span multiple 1 km grid-squares in the UK modelling outputs, with three of the sites > 1,000 ha and consisting of multiple component parts).

For the other four sites, where detailed feature maps were not available, the UKCEH Land Cover Map 2019 (Morton et al. 2020) was used to identify areas of deciduous woodland, as the closest proxy. To determine the suitability of this approach, i.e. using land cover as a proxy, a comparison between areas of H9180 (Tilio-Acerion) and/or H91A0 (Quercus petraea) woodland at Meirionnydd Oakwoods and Bat Sites SAC and deciduous woodland in LCM2019 was also carried out.

2.1.2 Sub-site selection

Some of the sites selected for the project are large and expansive, covering large areas of Wales. This makes identifying local emission sources difficult as agricultural activities and associated emissions can be highly spatially variable. Therefore, in order to provide meaningful guidance on likely key sources of atmospheric N input to sites, smaller sub-sites have been selected and characterised, in addition to the complete sites. Sub-sites were selected based on local knowledge (NRW, pers. comm.), primarily about lichen-rich areas, but also areas with emission sources nearby (identified through Google imagery and confirmed with local knowledge).

2.1.3 Current & future atmospheric N input (NH_3 concentrations and N deposition) at each site.

Current and predicted future rates of atmospheric N input were assessed for each site and the surrounding areas. Estimates of NH_3 concentrations and N deposition were taken from recent modelling work carried out under the JNCC project Nitrogen Futures (Dragosits et al. 2020). Modelling estimates for the “present day” were for the year 2017 and future estimates were for the year 2030. The 2030 estimates assume that commitments under the National Emission Ceilings Regulations (NECR) 2018 are met (further details are provided in Dragosits et al. 2020). The 2030 scenario estimate was used here to assess future mitigation strategies and their effect on the habitats/designated features, at a 1 km grid resolution.

These 1 x 1 km gridded estimates were used to estimate minimum, maximum and average estimates of NH_3 concentrations and N deposition to deciduous woodland and other semi-natural features at each site. Maps of NH_3 concentrations and N deposition were also produced to assess the spatial variability of inputs across each site.

2.1.4 Identification of the main sources of N deposition to each site.

The latest available source attribution data (for the year 2018, UKCEH data available to view on http://www.apis.ac.uk/app) were used to identify the main sources of atmospheric N deposition received by each site. The 5 km grid resolution dataset provides estimates of N deposition produced by 23 major UK point sources, international emissions (international shipping; and separate emissions for Republic of Ireland and the other EU countries, respectively) and 17 area emission source categories (including livestock, fertiliser and road transport), which are estimated separately for each UK country (i.e. Wales, England, Northern Ireland and Scotland).

Previous estimates of sources of N deposition to designated sites in Wales (made under the AAANIS project; Carnell *et al.* 2015) used an earlier source attribution dataset for the year 2005. The most recent data (for the year 2018) also provide additional details on the origin of N deposition sources by separating N species depositing mainly locally (NH_3, NO_2) from those that tend to disperse further and are deposited regionally or even internationally through long-range transport (i.e. NH_4^+, HNO_3), although this is a gradient rather than a clear distinction. Distinguishing between these types of chemical species provided an indication of whether the sources of N deposition received by a site were more likely of local origin or from further afield. The distinction between local and regional sources of N deposition was critical for the selection of suitable N mitigation measures, as it indicated whether targeting local sources was appropriate or whether wider initiatives to target N regionally, nationally or internationally would be required.

For each site, the main sources of N deposition were summarised and their relative and absolute contributions to total N deposition received by the site were estimated. The major sources of locally depositing N species (NH_3, NO_2) were also estimated. Local spatially targeted measures have been shown to be effective, compared with regionally depositing species (i.e. NH_4^+, HNO_3) that require more widespread mitigation efforts.

A desk based study of Google aerial and satellite imagery was carried out to identify potential emission sources within 1 km of site boundaries. This exercise helped identify likely local emission sources surrounding sites, which may suitable for spatially targeted mitigation.

2.1.5 Current & future agricultural emission density estimates.

The main sources of agricultural NH_3 emissions surrounding each site were estimated and results were calculated separately by source sector, where possible. The high-resolution holding-level agricultural statistics are subject to strict data licencing restrictions, therefore emissions from agricultural sectors associated with fewer than 5 agricultural holdings for any concentric zone around a site (or sub-site) need to be merged with other emission sources to comply with data agreements in place, until non-disclosive outputs can be achieved. The data were summarised to show the relative contribution of the following source types; dairy cattle, other cattle (i.e. beef), sheep, pigs, poultry, grass, crops and other sources. Where source sectors were disclosive (e.g. insufficient number of holdings with dairy cattle present), these were amalgamated into 'other sources' (containing a minimum of 5 holdings).

The agricultural density estimates calculated for this work are an improvement to estimates made under N Futures, as the Welsh Government was able to provide detailed coordinates of holding locations (rather than allocation to parish boundary centroids). This detailed location information enabled improved estimates of where emission sources are in relation to sites, for 1 km concentric buffer zones from the boundary of each site, to a maximum distance of 10 km.

2.2 Phase 2: identification of suitable measures to bring sites into good condition with regard to nitrogen

2.2.1 Off-site measures to reduce emissions

Emission sources close to the boundary of designated sites, such as livestock houses, or slurry stores, were identified using aerial and satellite imagery. These were shared with staff at the Snowdonia National Park Authority and the wider project steering group, for any local knowledge. Draft potential measures were identified for these local NH_3 sources, with likely emission savings estimated.

2.2.2 On-site restoration measures

A literature review was carried out to identify potentially suitable measures for the two woodland habitats studied. This was based on previous reviews conducted for JNCC as well as the extensive wider literature on N impacts in woodlands. However, literature on interactions of management and N deposition in most habitats, including woodlands, is rather limited.

2.3 Phase 3: Guidance on implementing SNAPs and an approach auditing effects on atmospheric N input to and recovery of sites

Phase 3 was carried out in two steps:

- A summary assessment of current and likely future threats to each site from atmospheric N input (from Phase 1) and recommendations for on-and off-site measures to mitigate effects on sensitive habitats and species (from Phase 2).
- A draft framework suggesting potential approaches and mechanisms for implementing SNAPs, including the processes required for a future ecological audit, to be carried out following implementation of SNAP measures. This audit is required to evaluate whether the expected reduction in atmospheric N input from local

measures has been achieved, and whether, together with the on-site habitat measures, has led to the achievement of the conservation objectives for the site.

3 Results

3.1 Identification of key nitrogen threats to sites and their features

3.1.1 Location and sensitivity of habitats within sites.

A comparison between areas of H9180 (*Tilio-Acerion*) and/or H91A0 (*Quercus petraea*) woodland at Meirionnydd Oakwoods and Bat Sites SAC and the more generic "deciduous woodland" category in LCM2019 was carried out. Figure 1 shows a good agreement between areas of H9180 and H91A0 (in the feature map) and areas of broadleaved woodland in LCM2019. A spatial comparison is also presented in Figure 2. Although it is not possible to distinguish between different deciduous woodland types in LCM2019, being able to identify areas of woodland allows improved assessments of critical loads and levels exceedance, especially for sites that span a larger geographic area.

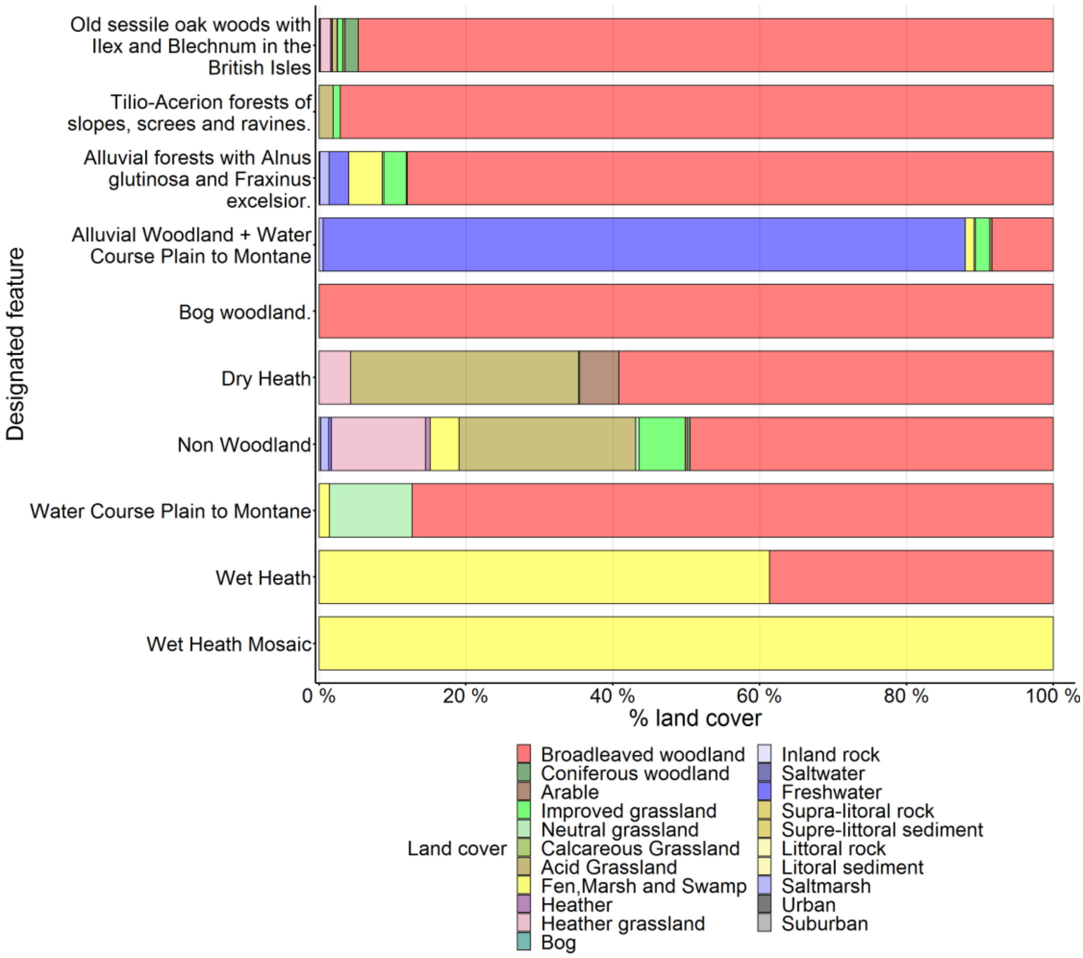

Figure 1: *A comparison between designated features for Meirionnydd Oakwoods and Bat Sites SAC (from a detailed feature map, y-axis) and UKCEH's Land Cover Map 2019 (Morton et al. 2020, fill colour).*

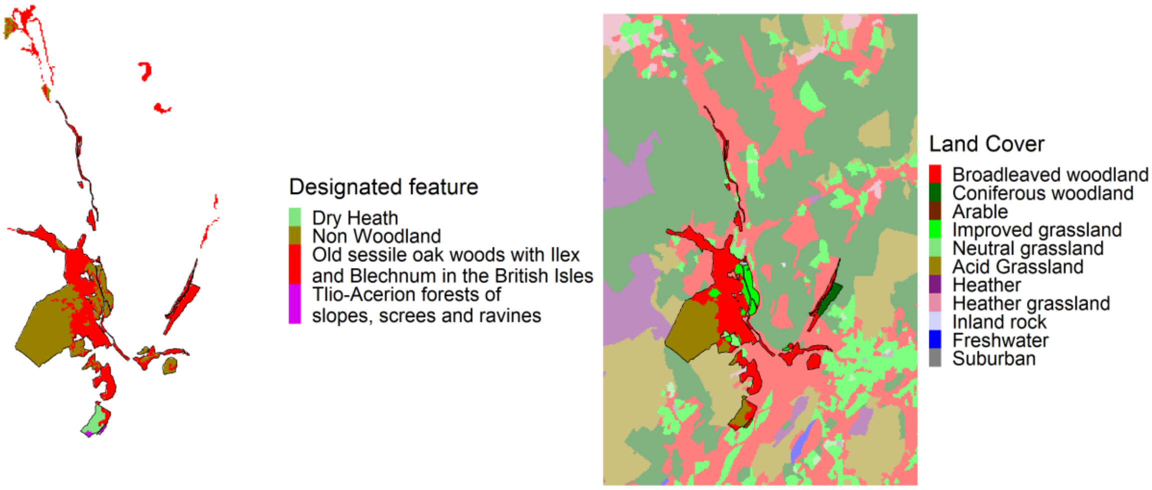

Figure 2: *Spatial distribution of designated features at the Parc Dolmelynllyn area of Meirionnydd Oak Woodlands versus land cover in the LCM2019.*

The UKCEH Land Cover Map 2019 (Morton et al. 2020) was used to identify areas of deciduous woodland for the four remaining sites, where feature maps were not available (Table 2). This is likely to be an overestimate of the presence of H9180 and H91A0, but provides a proxy and excludes areas of the sites which are mainly occupied by other (non-woodland) habitat types such as low-growing heaths, grasslands, montane vegetation etc. For example, 99 % of the southern part of Eryri/Snowdonia SAC is not covered by woodland, in contrast to Coed Cwm Einion, which has 96% woodland cover.

Table 2: *Site area and proportion of deciduous woodland (derived from LCM2019) at each of the study sites*

Site	Site Area (ha)	Deciduous woodland %	All other areas %
Coedydd Derw a Safleoedd Ystlumod Meirion / Meirionnydd Oakwoods and Bat Sites	2,813	84.5	15.5
Cwm Doethie / Mynydd Mallaen	4,122	12.1	87.9
Coed Cwm Einion	21	95.9	4.1
Southern part of Eryri / Snowdonia	3,615	0.9	99.1
Coetiroedd Cwm Elan / Elan Valley Woodlands	439	32.9	67.1

3.1.2 Sub site selection

Air quality experts at Natural Resources Wales who have extensive knowledge of the study sites suggested the following sub-sites:

- **Parc Dolmelynllyn area of Meirionnydd Oakwoods SAC**: The area of Parc Dolmelynllyn was chosen as it represents a relatively clean section of the larger Meirionnydd Oakwoods and Bat Sites SAC. This sub-site contains one of the richest assemblages of N-sensitive *Lobarion* lichen species in Wales and is located in an extensive agricultural landscape with few local emission sources.

- **Hafod Garegog area of Meirionnydd Oakwoods and Bat Sites SAC:** This site was selected as it provides a contrast to the Parc Dolmelynllyn sub-site of Meirionnydd Oakwoods and Bat Sites SAC (described above). Hafod Garegog contains very sensitive lichen species but is located near the intensively farmed Prenteg flatlands. The sub-site is situated near cattle farms and slurry stores are visible in satellite images within ~2km of the site.
- **Llyn Gwynant area of Eryri/Snowdonia SAC:** This sub-site was selected because it contains some of the largest selections of woodland lichen species in Eryri/Snowdonia SAC and is situated on the opposite side of the valley from the northernmost units of Meirionnydd Oakwoods and Bat Sites SAC.
- **Coed Cwm Einion SAC**: At 21 hectares, Coed Cwm Einion is one of the smaller SACs assessed, and therefore the whole site was included in the assessment.
- **Gwenffrwd Woods at Cwm Doethie – Mynydd Mallaen SAC**: Gwenffrwd Woods was selected as this is probably the most lichen-rich woodland in the wider SAC.
- **Coetiroedd Cwm Elan / Elan Valley Woodlands SAC** at **Carn Gafallt SN938651** is exceptionally lichen rich but within ~2 km from a livestock unit identified by local satellite imagery.

3.1.3 Current & future atmospheric N input (NH_3 concentrations and N deposition) at each site.

Average rates of N deposition (to woodland features) to the five sites were estimated to range between 15.7 and 21 kg N ha^{-1} yr^{-1} (Table 3). These current (2017) rates of N deposition are in exceedance of the (mapping) critical load for the sessile oak woodland features (H91A0) of 10 kg N ha^{-1} yr^{-1} (H91A0, sessile oak) and for some sites above the critical load for Tilio-Acerion woodlands of 17.5 kg N ha^{-1} yr^{-1} (H9180). N deposition rates are estimated to decline at all sites under modelled 2030 N emissions, under the premise that the National Emission Ceilings Regulations (NECR) targets are met (details of the emission scenario can be found in Dragosits *et al.* 2020). Despite the projected reduction in N deposition across all sites, average deposition in 2030 was still in exceedance of the most sensitive critical load for the woodland features at all sites (with critical loads exceeded by ~9 - 10 kg N ha^{-1} yr^{-1}).

Coetiroedd Cwm Elan / Elan Valley Woodlands has the highest average N deposition of the five sites of 21 kg N ha^{-1} yr^{-1} and an average exceedance of 11 kg N ha^{-1} yr^{-1}, (a slight reduction to 19.5 kg N ha^{-1} yr^{-1} is estimated by 2030). Despite having the lowest average estimated N deposition of the five sites, the sessile oak woodlands at Coed Cwm Einion are estimated to be in exceedance of the critical load by 5.7 kg N ha^{-1} yr^{-1}. The size of the five SACs varies, at small sites such as Coed Cwm Einion (21 ha) the 1 x 1km grid resolution of the N deposition model output is relatively coarse and the 1km grid estimate may not reflect conditions at the site. At larger sites such as Cwm Doethie / Mynydd Mallaen the rate of N deposition may vary much more widely across the wider site area. Sub-sites have been selected at these larger sites in a more targeted way, to assess N deposition rates and identify local sources for key areas.

Table 3: *Area-weighted average N deposition to deciduous woodland at each of the study sites in 2017 and 2030, estimated using FRAME model output from the Nitrogen Futures project. The range in N deposition received by each site is shown in brackets.*

		Mean N Deposition (kg N ha^{-1} yr^{-1})		
Site	**Area (ha)**	**2017**	**2030**	***Change (%)***
Coedydd Derw a Safleoedd Ystlumod Meirion / Meirionnydd Oakwoods and Bat Sites	2,813	17.6 (13.6 – 24.6)	16.4 (12.6 - 22.9)	-7%
Cwm Doethie / Mynydd Mallaen	4,122	20.0 (18.4 - 21.2)	18.4 (16.9 - 19.5)	-8%
Coed Cwm Einion	21	15.7 (14.7 – 16.0)	14.5 (13.6 - 14.8)	-8%
Southern part of Eryri / Snowdonia	3,613	24.4 (20.0 – 28.4)	22.7 (18.6 – 26.5)	-7%
Coetiroedd Cwm Elan / Elan Valley Woodlands	439	21.0 (19.2 - 23.5)	19.5 (17.8 - 21.8)	7%

Of the six sub-sites selected, two had the same mean N deposition as the wider site. The Llyn Gwynant area of Eryri/Snowdonia SAC and Gwenffrwd Woods in Cwm Doethie - Mynydd Mallaen SAC had lower rates of atmospheric N deposition than the whole site. Hafod Garegog and Coetiroedd Cwm Elan SAC saw a higher estimated N deposition rate than the whole site average.

Table 4: *Area-weighted average 2017 N deposition to deciduous woodland at each study sites and selected sub-sites (FRAME model output from the Nitrogen Futures project). Range in N deposition received by each site is shown in brackets.*

Site Name		Mean N Deposition (kg N ha^{-1} yr^{-1})	
Wider site	**Sub-site**	**Wider site**	**Sub-site**
Meirionnydd Oakwoods SAC	Parc Dolmelynllyn	17.6 (13.6 – 24.6)	17.6 (17.2 – 19.4)
Meirionnydd Oakwoods SAC	Hafod Garegog	17.6 (13.6 – 24.6)	17.8 (14.0 – 19.3)
Southern part of Eryri/Snowdonia SAC	Llyn Gwynant	24.4 (20.0 – 28.4)	23.8 (22.2 – 24.8)
Coed Cwm Einion	N/A	15.7 (14.7 – 16.0)	15.7 (14.7 – 16.0)
Cwm Doethie – Mynydd Mallaen	Gwenffrwd Woods	20.0 (18.4 - 21.2)	19.7 (18.8 – 20.4)
Coetiroedd Cwm Elan/ Elan Valley Woodlands	Carn Gafallt	21.0 (19.2 - 23.5)	21.2 (20.2 – 22.4)

Area-weighted mean NH_3 concentrations ranged between 0.5 and 0.7 μg m^{-3} at the five selected sites. The mean NH_3 concentration at all sites is relatively low compared to the rest of the UK and area-weighted mean concentrations are all estimated to be below the 1 μg CLe for lichens and bryophytes. However, some areas within sites do exceed this CLe. Ammonia concentrations are expected to decline slightly at all five sites by 2030, assuming NECR emissions targets are met. Estimated maximum concentrations will be most affected. Given these projections, all 1 km grid averages within sites are expected to decline to below the 1 μg CLe by 2030.

In reality, NH_3 concentrations are more spatially variable than is represented by the 1 km x 1 km grid resolution. Steep gradients in NH_3 are observed near agricultural emission sources, and woodlands can intercept NH_3 and significantly reduce concentrations and loads at downwind locations. At all sites, there are likely to be areas with higher local concentrations than shown by the area-weighted averages shown in Table 5.

Table 5: *Area-weighted average (2017 and 2030) NH_3 concentrations at each study site, estimated using model outputs from the Nitrogen Futures project.*

	Area-weighted mean (min/max) NH_3 concentrations ($\mu g\ m^{-3}$)		
Site	**2017**	**2030**	***Change***
Coedydd Derw a Safleoedd Ystlumod Meirion / Meirionnydd Oakwoods and Bat Sites	0.5 (0.4 – 1.2)	0.5 (0.4 – 0.9)	*- 0.02*
Cwm Doethie / Mynydd Mallaen	0.7 (0.5 – 1.0)	0.6 (0.5 – 0.9)	- 0.04
Coed Cwm Einion	0.5 (0.5 – 0.6)	0.5 (0.5 – 0.5)	- 0.01
Southern part of Eryri / Snowdonia	0.5 (0.5 - 1.6)	0.5 (0.5 - 1.6)	- 0.01
Coetiroedd Cwm Elan / Elan Valley Woodlands	0.6 (0.5 - 1)	0.6 (0.5 -0.9)	- 0.02

Average area-weighted NH_3 concentrations were also estimated specifically for the areas of broadleaved woodland at sites. These are presented in the site profiles (Annex 1). These woodland specific estimates were similar to the site average (i.e. for all land cover types), this is most probably due to the ammonia concentrations being estimated at a 1 km x 1km grid resolution, which is relatively coarse compared with the 25m x 25m resolution of the land cover dataset (LCM 2019). For larger sites that span a steep concentration gradient, estimating NH_3 concentrations for the woodland areas only may be a useful exercise, especially if the areas are concentrated in a small area of a site.

Estimated area-weighted average ammonia concentrations for the selected sub-sites were similar to the wider SACs, with concentrations ranging from 0.4 – 0.7 $\mu g\ m^{-3}$. The Parc Dolmelynllyn sub-site was selected as a relatively clean section of Meirionnydd Oakwoods and Bat Sites SAC and indeed has a lower estimated average ammonia concentration than the wider site average. Despite having some of the most sensitive lichen species, the Hafod Garegog sub-site (of Meirionnydd Oakwoods and Bat Sites SAC) has a higher mean NH_3 concentration at 0.6 $\mu g\ m^{-3}$, and some areas with (1 km x 1 km grid resolution) modelled concentrations of 1.2 $\mu g\ m^{-3}$.

Table 6: *Mean area-weighted NH_3 concentration at each study site and selected sub-sites from 2017 Nitrogen Futures model.*

Site Name		Area-weighted mean (min/max) NH_3 concentrations ($\mu g\ m^{-3}$)	
Wider site	**Sub-site**	**Whole site**	**Sub-site**
Meirionnydd Oakwoods SAC	Parc Dolmelynllyn	0.5 (0.4 – 1.2)	0.4 (0.4 -0.5)
Meirionnydd Oakwoods SAC	Hafod Garegog	0.5 (0.4 – 1.2)	0.6 (0.4 – 1.2)
Southern part of Eryri/Snowdonia SAC	Llyn Gwynant	0.5 (0.5 - 1.6)	0.5 (0.5 – 0.6)
Coed Cwm Einion	N/A	0.5 (0.5 – 0.6)	0.5 (0.5 – 0.6)
Cwm Doethie – Mynydd Mallaen	Gwenffrwd Woods	0.7 (0.5 – 1.0)	0.7 (0.6 – 0.7)
Coetiroedd Cwm Elan/ Elan Valley Woodlands	Carn Gafallt	0.6 (0.5 – 0.7)	0.6 (0.5 – 0.7)

3.1.4 Identification of the main sources of N deposition to each site

The latest source attribution dataset for the emission year 2018 indicates that the main source of N deposition for the wider Coed Cwm Einion site is livestock emissions from within Wales (~35 % of total N deposition with ~9 % from livestock sources in the rest of the UK), followed by European emission sources (~10.3 %,

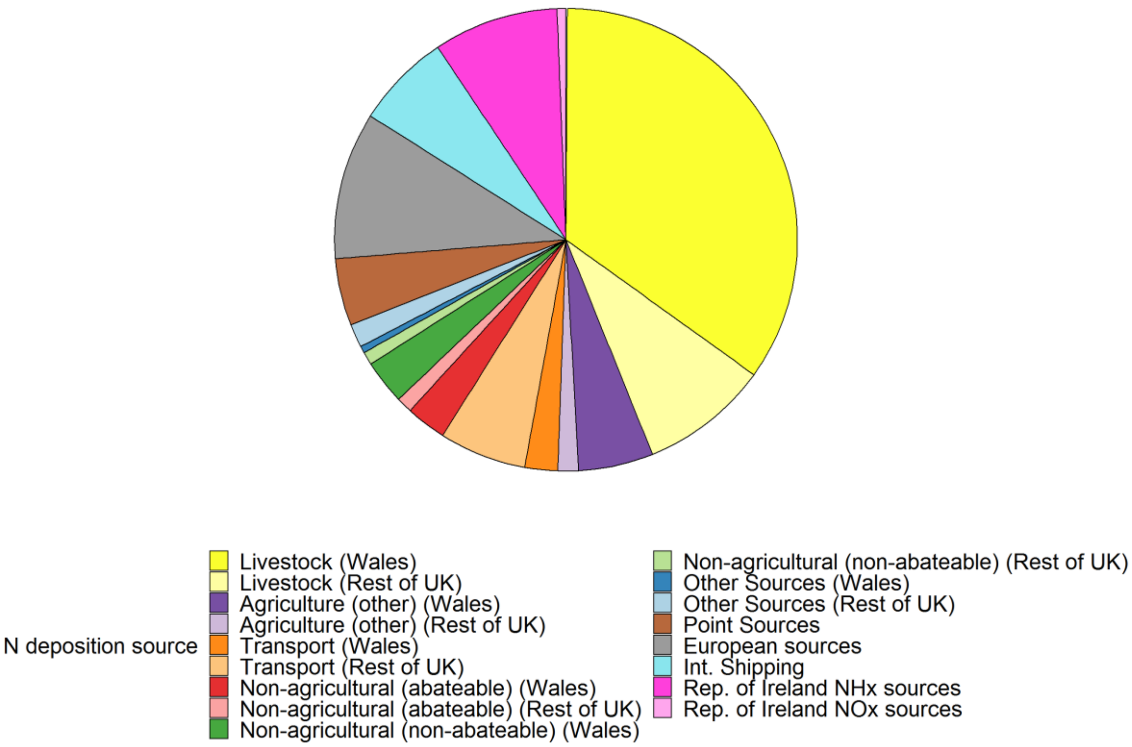

Figure 3). In contrast, the southern section of Eryri/Snowdonia receives most of its N deposition from European (non-Ireland) (17.4%) and Republic of Ireland (11.0 %) sources and a smaller fraction from livestock in Wales (19.2 %). A substantial fraction of the N deposition received by this site is from sources outside of Wales, with 13.3 % of N deposition from livestock sources in the rest of the UK 19.2 % within Wales.

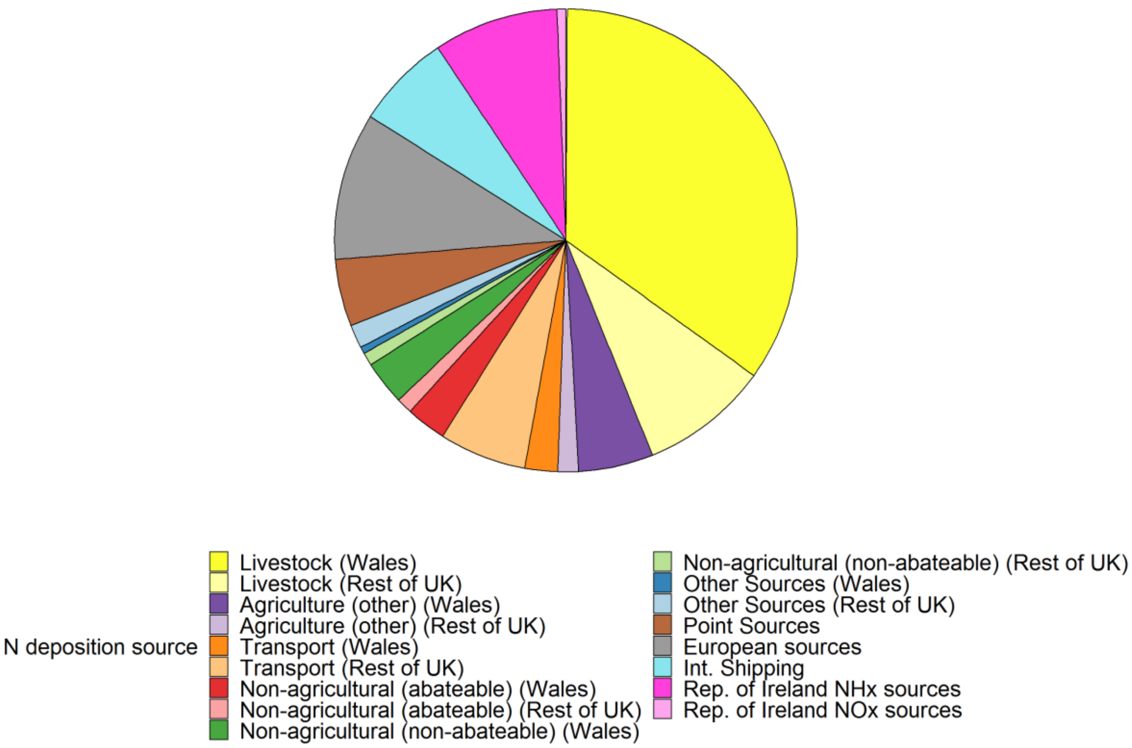

Figure 3: *2018 source attribution of total N deposition to woodland features for Coed Cwm Einion. Area-weighted average N deposition (2018, 5 x 5 km grid resolution) for Coed Cwm Einion was 13.1 kg N ha^{-1} yr^{-1}.*

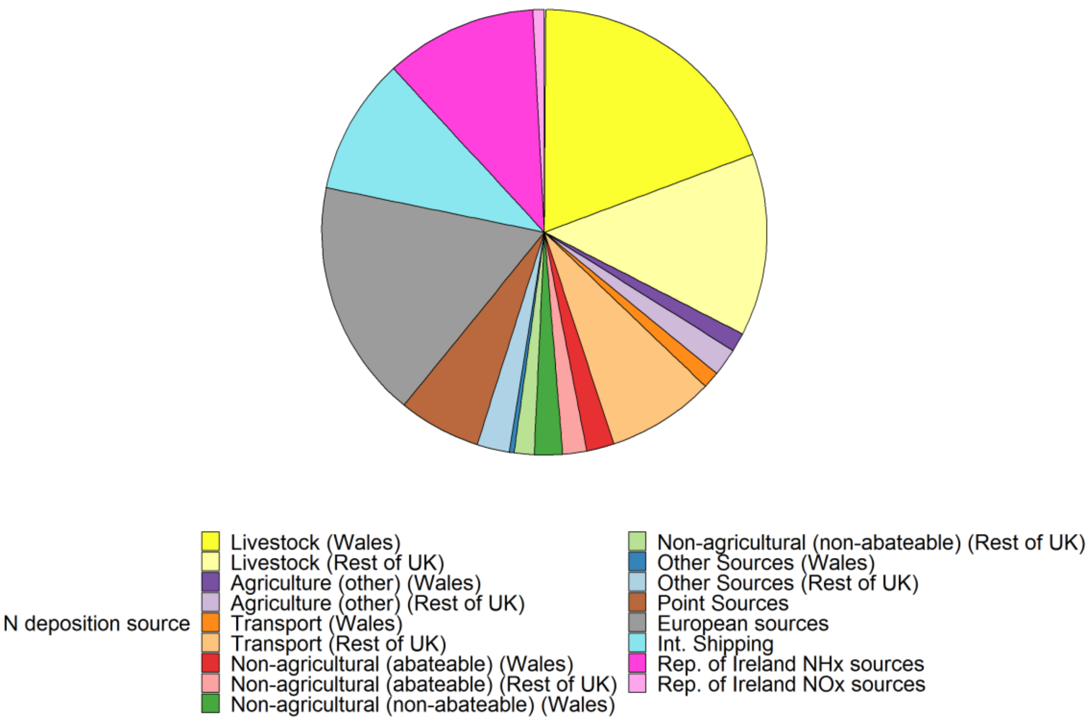

Figure 4: *2018 source attribution of total N deposition to woodland features for Southern part of Eryri/Snowdonia. Area-weighted average N deposition for Eryri/Snowdonia was 17.4 kg N ha^{-1} yr^{-1}.*

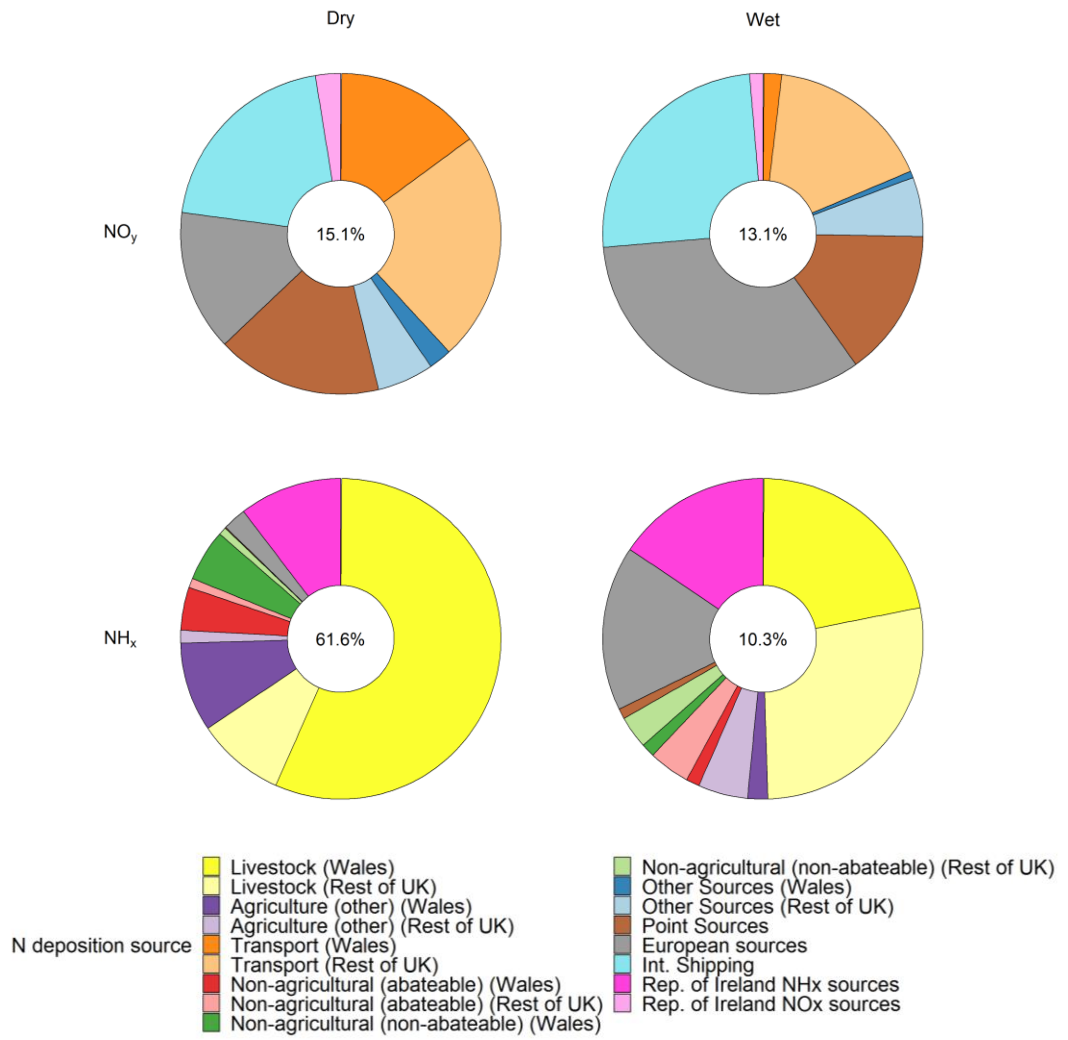

Figure 5: *Woodland source attribution for Coed Cwm Einion, showing Oxidised vs Reduced & Wet vs Dry N deposition. % values indicate contribution to total N deposition.*

Most of the N deposition received by Coed Cwm Einion is in the form of dry N deposition (which is typically associated with local sources compared with wet deposition). Over half (61.6 %) of the total N deposition to woodland features is in the form of dry NH_x deposition, mainly from livestock sources (Figure 5). The majority of dry NH_x deposition received by Coed Cwm Einion is attributed to livestock within Wales, suggesting targeted mitigation of agricultural sources near the site could have a significant impact on N deposition for the site.

In contrast, the southern part of Eryri/Snowdonia receives a higher proportion of wet deposition (to woodland features, 55.1 % of total N deposition, Figure 6). Wet deposition is typically associated with regional/long range transport and suggests local measures may be less effective at this site. The largest source of the wet deposition received by the site is from European sources and International shipping, with smaller components from within Wales.

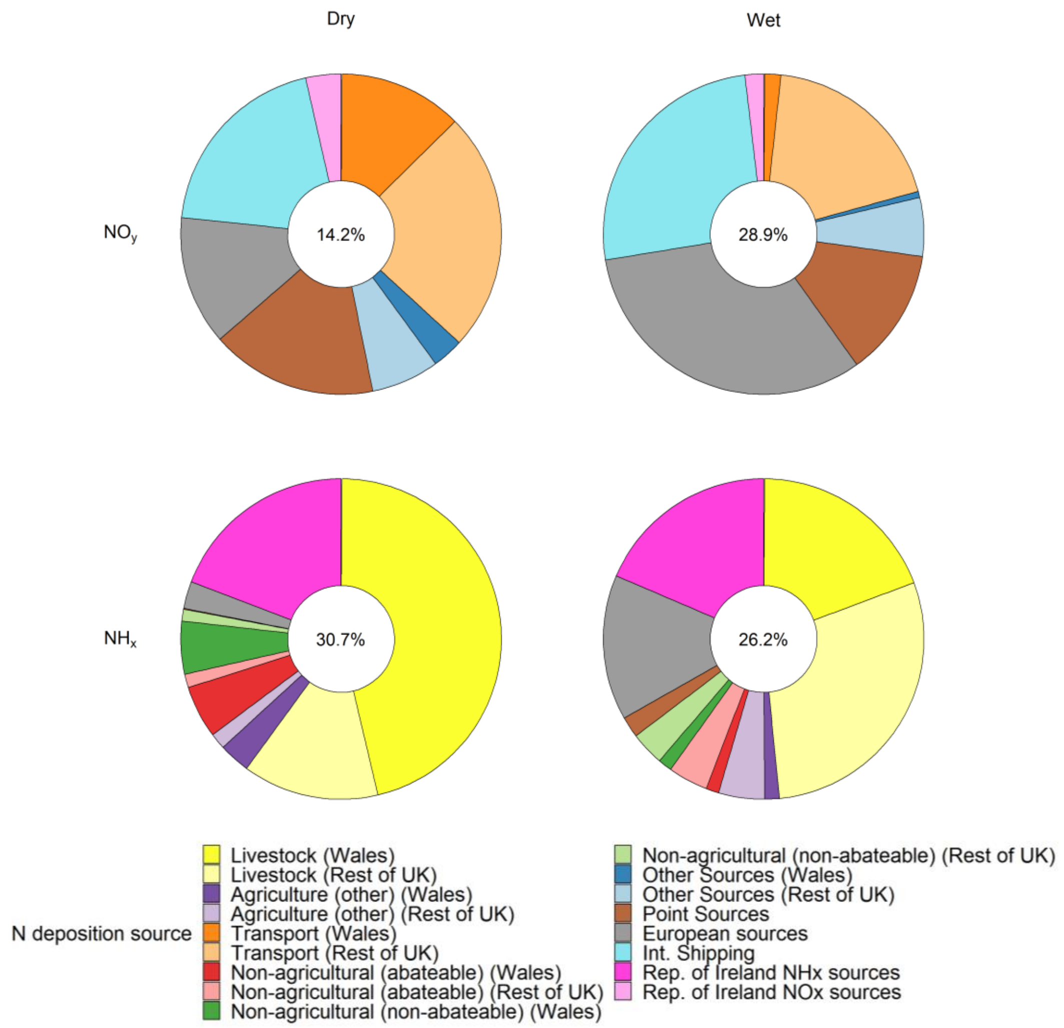

Figure 6: *Woodland vegetation source attribution for Snowdonia/Eryri, showing oxidised vs reduced & wet vs dry N deposition. % values indicate contribution to total N deposition.*

3.1.5 Agricultural emission density estimates

Estimated agricultural emission densities surrounding all sites was relatively low compared to other parts of the UK (e.g. sites selected in Dragosits et al. 2020), with average emission densities within 2 km of site boundaries ranging from 2.2 – 4.6 kg NH_3 ha^{-1} yr^{-1} (Table 7). There were fewer than five pig units and fewer than five poultry units within a 2 km buffer zone of all sites, so any emissions from pig and poultry units were included with the category “Other sources”, with a combined number of at least five units in total.

Table 7: *Average agricultural NH_3 emission density surrounding the five selected sites, with the proportion of emissions indicated for each agricultural sector present, for a 2 km zone around each site. Emissions from fewer than 5 agricultural sources (inc. those from pig and poultry) have been aggregated into the category "other sources"*

Site Name	kg NH_3 ha^{-1} yr^{-1}	Beef	Dairy	Sheep	Horses, Goats & Farmed deer	Mineral Fertiliser	Other sources*
Eryri/ Snowdonia	3.0	33%	10%	36%	4%	12%	3%
Southern part of Eryri/Snowdonia	3.4	34%	-	41%	13%	12%	0%
Coedydd Derw a Safleoedd Ystlumod Meirion/ Meirionnydd Oakwoods and Bat Sites	2.5	36%	8%	37%	1%	17%	1%
Coed Cwm Einion	2.3	-	-	35%	-	15%	50%
Cwm Doethie - Mynydd Mallaen	1.8	28%	1%	47%	0%	22%	1%
Coetiroedd Cwm Elan/ Elan Valley Woodlands	2.4	22%	-	61%	2%	14%	1%

Most of the sites are characterised by a high proportion of emissions associated with beef cattle and sheep. The Coed Cwm Einion site has a high proportion of emissions from other sources (50 %) within 2km of the site boundary. It may be hypothesised that this could be associated with dairy and beef farming, with beef farming no longer disclosive beyond 2 km and dairy farming split out beyond the 5 km zone (Figure 7). While it may appear that there are no pigs, poultry, horses, goats or deer present at the site, these are however just aggregated with any disclosive sectors into "other emissions" to prevent non-disclosive sectors being shown. "Other sources" in the >6km zones are extremely small beyond 5 km distance (~0.25%) so are difficult to see in Figure 7.

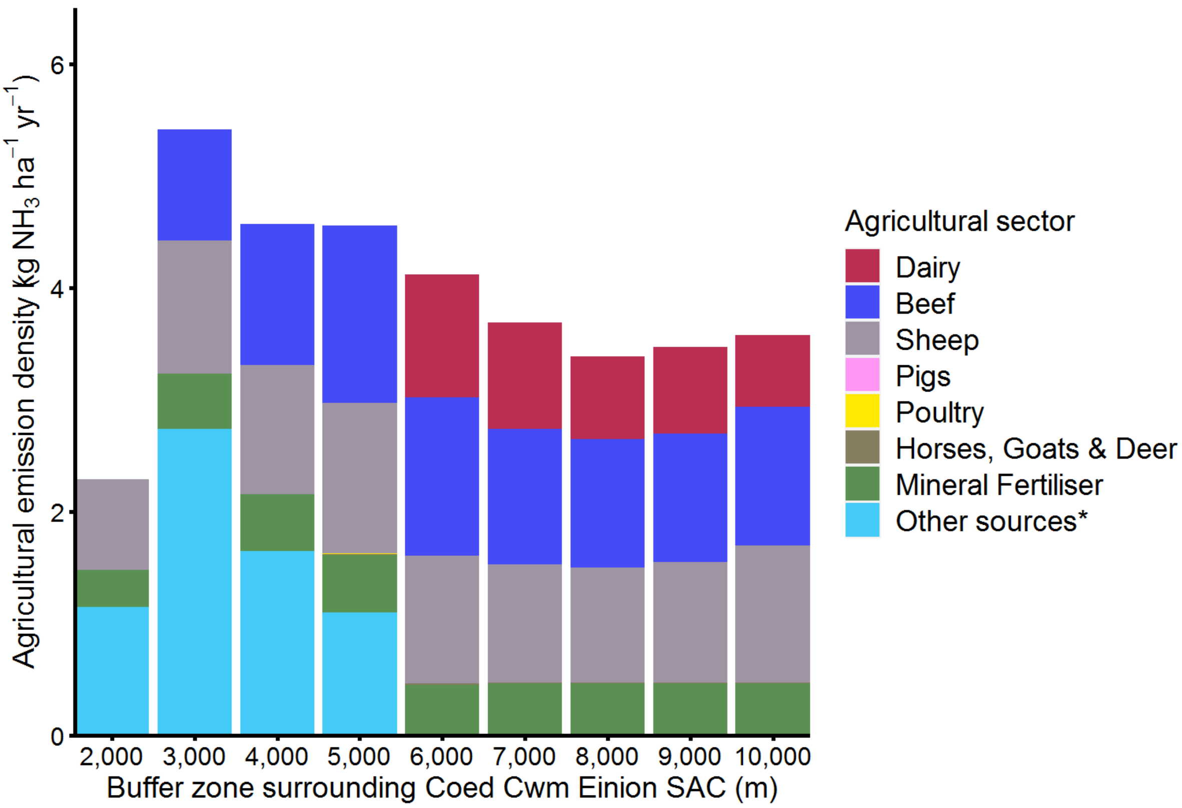

Figure 7: *2017 Agricultural emission density estimates for Coed Cwm Einion SAC in concentric rings from the site boundary to distances of 2 – 10km (i.e. 0 – 2km, 0 – 3km …). The category 'other sources' refers to any emissions sources which are disclosive, i.e. data points from less than five agricultural holdings.*

Although the average agricultural emission densities surrounding the wider sites are relatively low in the context of the rest of the UK, it is important to note that this is an average for the entire 2 km buffer zone surrounding the entire selected sites (some of which are very large) and some areas of sites may be within emissions hotspots.

The Hafod Garegog sub-site of Meirionnydd Oakwoods and Bat Sites SAC has a higher average agricultural emissions density than the Parc Dolmelynllyn (Table 8), which is reflected in higher modelled NH_3 concentrations.

Table 8: *Average agricultural NH_3 emission density surrounding the selected sub-sites sites, with the proportion of emissions indicated for each agricultural sector present, for a 2 km zone around each sub-site. Emissions from fewer than five agricultural sources have been aggregated into the category "other sources"*

Wider site	Sub-site	kg NH_3 ha^{-1} yr^{-1}	Beef	Sheep	Poultry	Horses, Goats & Deer	Mineral Fertiliser	Other sources*
Meirionnydd Oakwoods SAC	Parc Dolmelynllyn	0.7	24%	46%	1%	-	29%	0%
Meirionnydd Oakwoods SAC	Hafod Garegog	3.8	36%	28%	0%	-	11%	25%
Southern part of Eryri/Snowdonia SAC	Llyn Gwynant	1.9	-	45%	-	-	12%	42%
Coed Cwm Einion		2.3	-	-	35%	-	-	15%
Cwm Doethie – Mynydd Mallaen	Gwenffrwd Woods	1.5	41%	-	-	0%	39%	20%
Coetiroedd Cwm Elan/ Elan Valley Woodlands	Carn Gafallt	2.3	21%	64%	0.3%	-	15%	0.2%

The average agricultural emission density at the Parc Dolmelynllyn sub-site was lowest within the 2km buffer zone (compared to the other concentric zones, see Figure 8), with higher densities further from the site driven by an increase in beef and sheep farming. In contrast, the total agricultural emission density at the Hafod Garegog site was highest within the 2km buffer zone and declined further from the site (Figure 9). The dominant emissions source surrounding Hafod Garegog was beef farming, followed by sheep. Emissions from dairy cattle become non-disclosive (with at least 5 dairy holdings) in the 9 km buffer zone and emissions from poultry and pigs are aggregated with "Other sources" to prevent any non-disclosive emissions being shown.

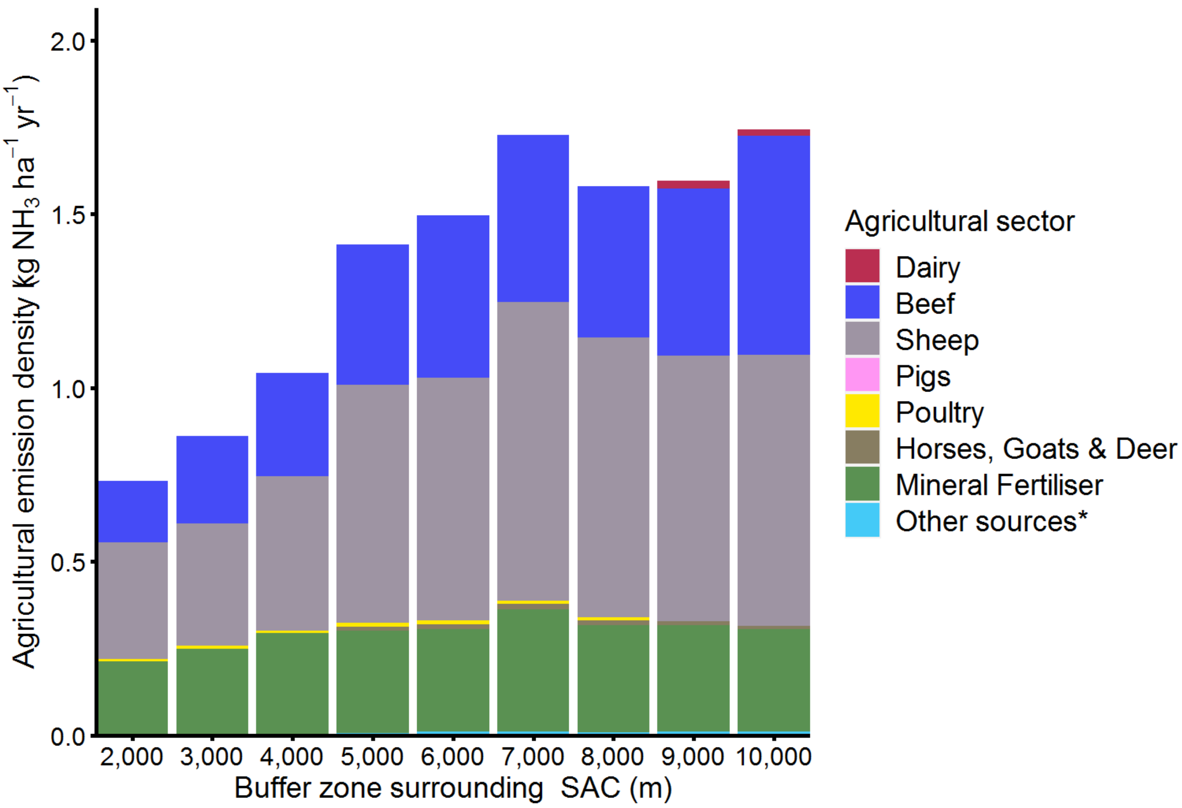

Figure 8: *2017 Agricultural emission density estimates for Parc Dolmelynllyn sub-site of Meirionnydd Oakwoods SAC in concentric rings from the site boundary to distances of 2 – 10km (i.e. 0 – 2km, 0 – 3km …). The category 'other sources' refers to any emissions sources which are disclosive (i.e. data points from less than five agricultural holdings) and are combined with at least one other category.*

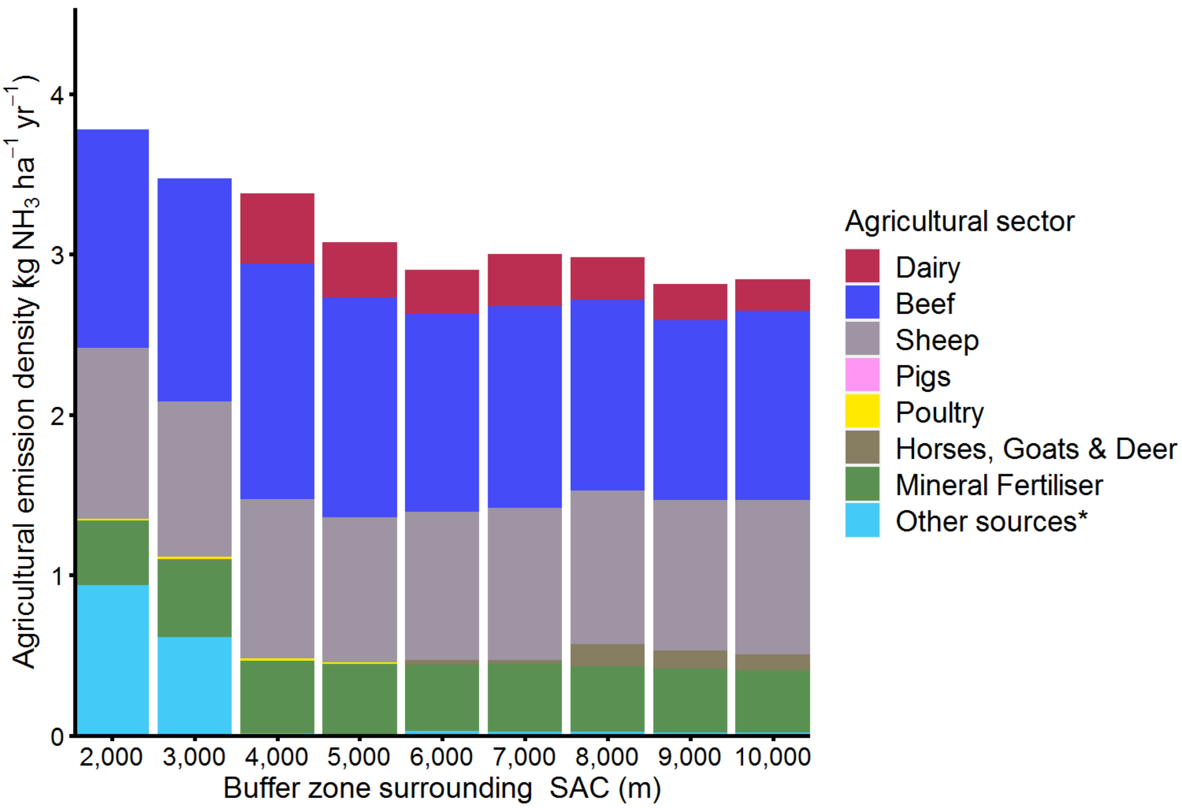

Figure 9: *2017 Agricultural emission density estimates for Hafod Garegog sub-site of Meirionnydd Oakwoods SAC in concentric rings from the site boundary to distances of 2 – 10km (i.e. 0 – 2km, 0 – 3km …). The category 'other sources' refers to any emissions sources which are disclosive (i.e. data points from less than five agricultural holdings).*

3.1.6 Site profiles

The analyses of the sites and sub-sites illustrated so far in Section 3.1 were summarised into individual site profiles. These can be found in Annex 1.

3.2 Phase 2a: Identification of suitable off-site mitigation measures to bring sites into good condition with regard to atmospheric nitrogen

For the smaller sites (Coetiroedd Cwm Elan, and some smaller sub-sites (Parc Dolmelynllyn and Hafod Garegog areas of Meirionnydd Oakwoods and Bat Sites SAC), a detailed search for potential local emission sources close to the site boundaries were carried out, using satellite photography and local knowledge, as described in Section 2.2.1. These are summarised below, with the likely effectiveness in reducing emissions with potentially suitable measures described.

Coetiroedd Cwm Elan / Elan Valley Woodlands SAC

Coedtiroedd Cwm Elan SAC appears to be situated in an area of extensive sheep and cattle farming. Bing aerial/satellite imagery of the area (Figure 10) shows two slurry lagoons (illustrated by yellow triangles) within ~1.2 km (east) and ~2km (northeast) of the site boundary. There also appears to be some grazing sheep and cattle and livestock houses within ~2 km of the sub-site boundary. The livestock house closest to the site boundary (on the south east side of the site) appears to have sheep grazing close to the site boundary. Although emissions from sheep grazing tend to be relatively modest (compared to other emission sources such as pig/poultry housing), emissions associated with grazing livestock on improved pasture, such as the application of synthetic fertiliser may lead to an emission hotspot locally, especially in fields so close to the site boundary. The rest of the sub-site is mainly surrounded by open water (to the west) and upland vegetation (to the north). The most obvious emission sources for local spatial targeting may therefore be the slurry lagoons identified ~1.5 km from the site. Installing a floating cover on to a slurry lagoon typically reduces NH_3 emissions by ~60% (assuming that there is no natural crust on the lagoon surface already).

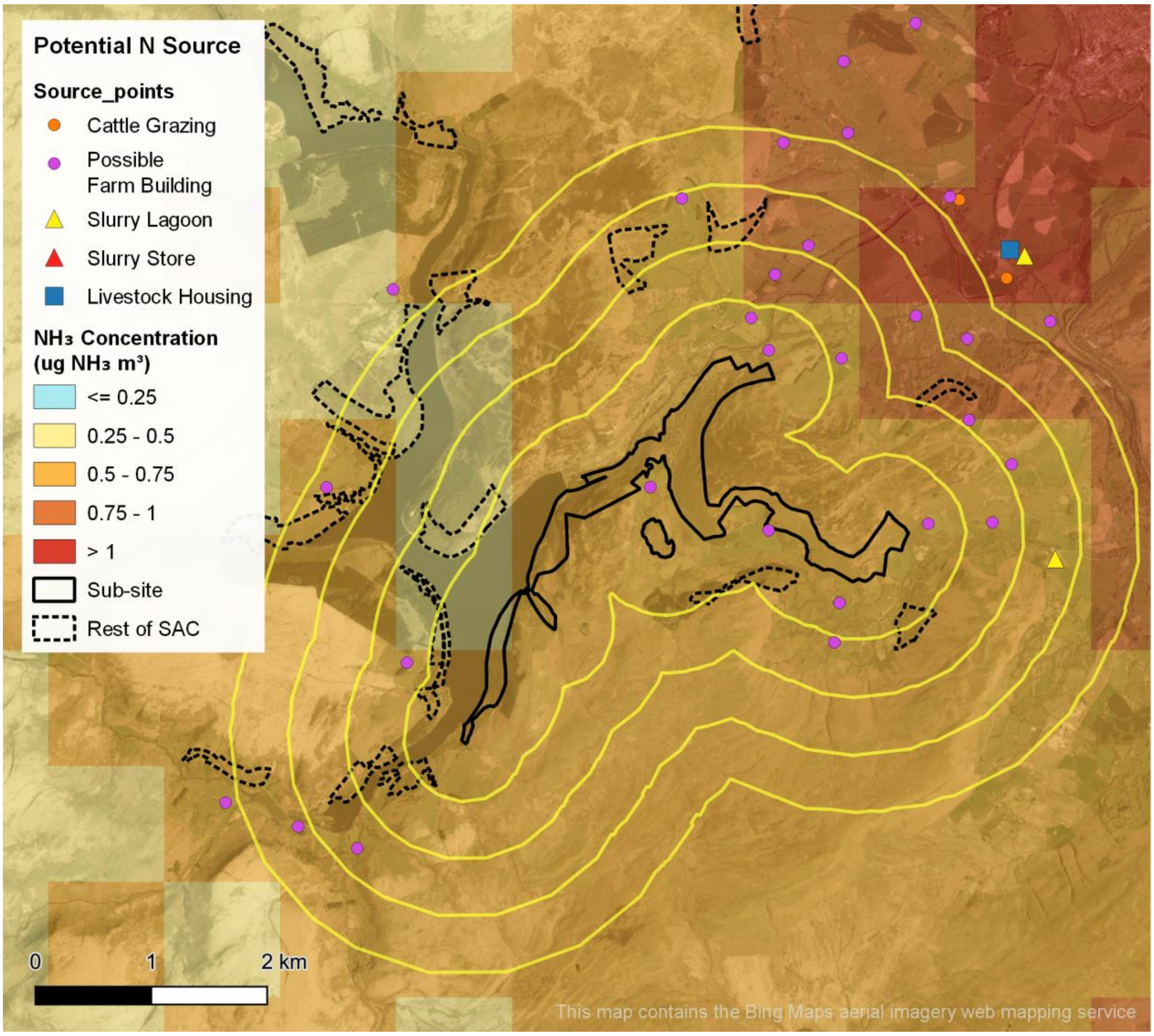

Figure 10: *Coetiroedd Cwm Elan SAC Carn Gafallt, potential N sources identified from Microsoft Bing Satellite Imagery and NH_3 concentration. 500m buffer zones around sub-site up to 2km shown in yellow.*

Parc Dolmelynllyn area

The Parc Dolmelynllyn sub-site of Meirionnydd Oakwoods and Bat Sites SAC appears to be situated in an area of extensive sheep and cattle farming and is mainly surrounded by woodland areas. Bing aerial/satellite imagery of the area (Figure 11) appears to show cattle grazing on a field of improved grassland within the site boundary. There is a beef cattle shed adjacent to the site boundary (illustrated by a blue square and identified by NRW pers comms). Although the Parc Dolmelynllyn site has a low estimated agricultural emission density (compared to the wider Meirionnydd Oakwoods and Bat Sites SAC, Table 8), cattle grazing and housing emissions so close to the site boundary may lead to emission hotspots and elevated NH_3 concentrations locally. One potential measure for this site may be to exclude cattle grazing from within and adjacent to the site boundary. Local knowledge however suggests that local cattle grazing does not appear to be an issue at this site as there are some outstandingly rich N-sensitive Lobarion lichen communities close to this cattle shed and cattle grazing has benefits for the local landscape. The site may also benefit from changes to the livestock and manure management practices at the farm steading (immediately adjacent to the site boundary) if it is not already adopting best practice. This may include improved cleaning and minimising overall extent of any outdoor yard areas used by cattle, keeping the house bedding dry by use and targeting of sufficient bedding, covering of manure heaps and spreading of manure to fields further away from the site boundary. This steading is likely to be beef housing (rather than dairy), and is therefore likely to be straw-bedded, which limits mitigation options.

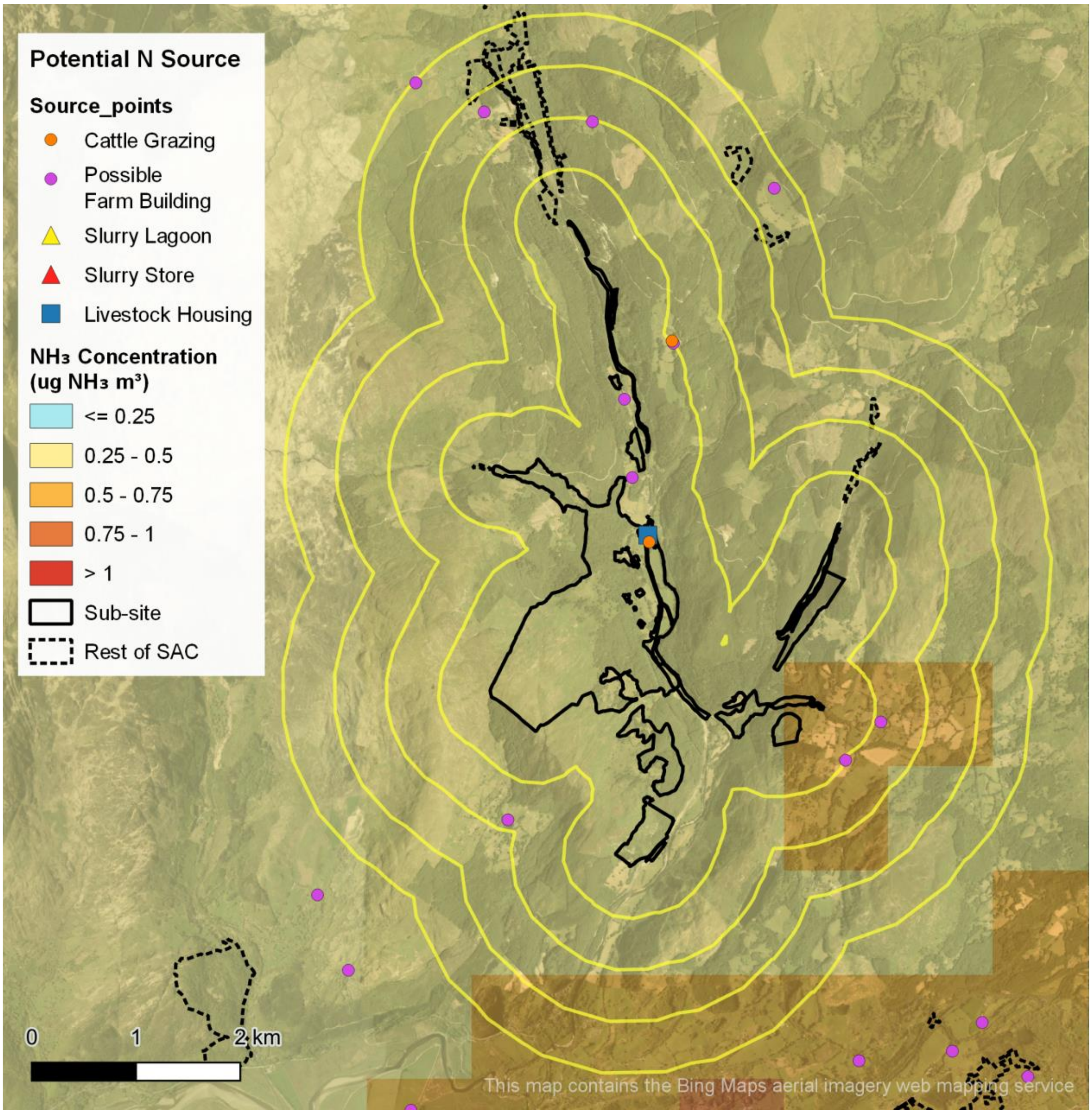

Figure 11: *Parc Dolmelynllyn area of Meirionnydd Oakwoods SAC with potential N sources identified from Microsoft Bing Satellite Imagery. 500m buffer zones around sub-site up to 2km shown in yellow.*

Hafod Garegog

The northern sections of Hafod Garegog (sub-site of Meirionnydd Oakwoods and Bat Sites SAC) is situated in an area of extensive upland agriculture, while the southern part of the sub-site is surrounded by improved grassland with grazing livestock. In contrast to Parc Dolmelynllyn, this sub-site seems to be fairly exposed, with fewer areas of woodland outside the site boundary helping to recapture nitrogen and reduce deposition to the site (Figure 12). This southern part of the sub-site has some of the highest NH_3 concentrations of the sites assessed in this study, with modelled 1km x 1km NH_3 concentrations of 1.2 µg NH_3 m^{-3} and therefore exceeding the 1 µg NH_3 m^{-3} critical level for lichens, mosses and bryophytes. Bing aerial/satellite imagery of the area appears to show an above ground slurry tank within ~750m of the site boundary (to the west of the southernmost part and indicated with a red triangle). This above ground tank appears to be uncovered (at the time the imagery was taken) and a potential off-site measure may therefore be to fit a fixed cover to the tank (if possible, depending on structural integrity) to lower its emissions by up to 80% (assuming there is no natural crust to the slurry store). A floating cover (giving 60% emission reduction) is another option if the tank structural integrity is insufficient for installation of a rigid cover. As with the Parc Dolmelynllyn site, improving the manure management systems of the livestock houses surrounding the site may also lead to reduced NH_3 emissions, depending on the systems currently used. Examples include minimising yard and housing floor of areas fouled by livestock excreta, covering manure heaps,

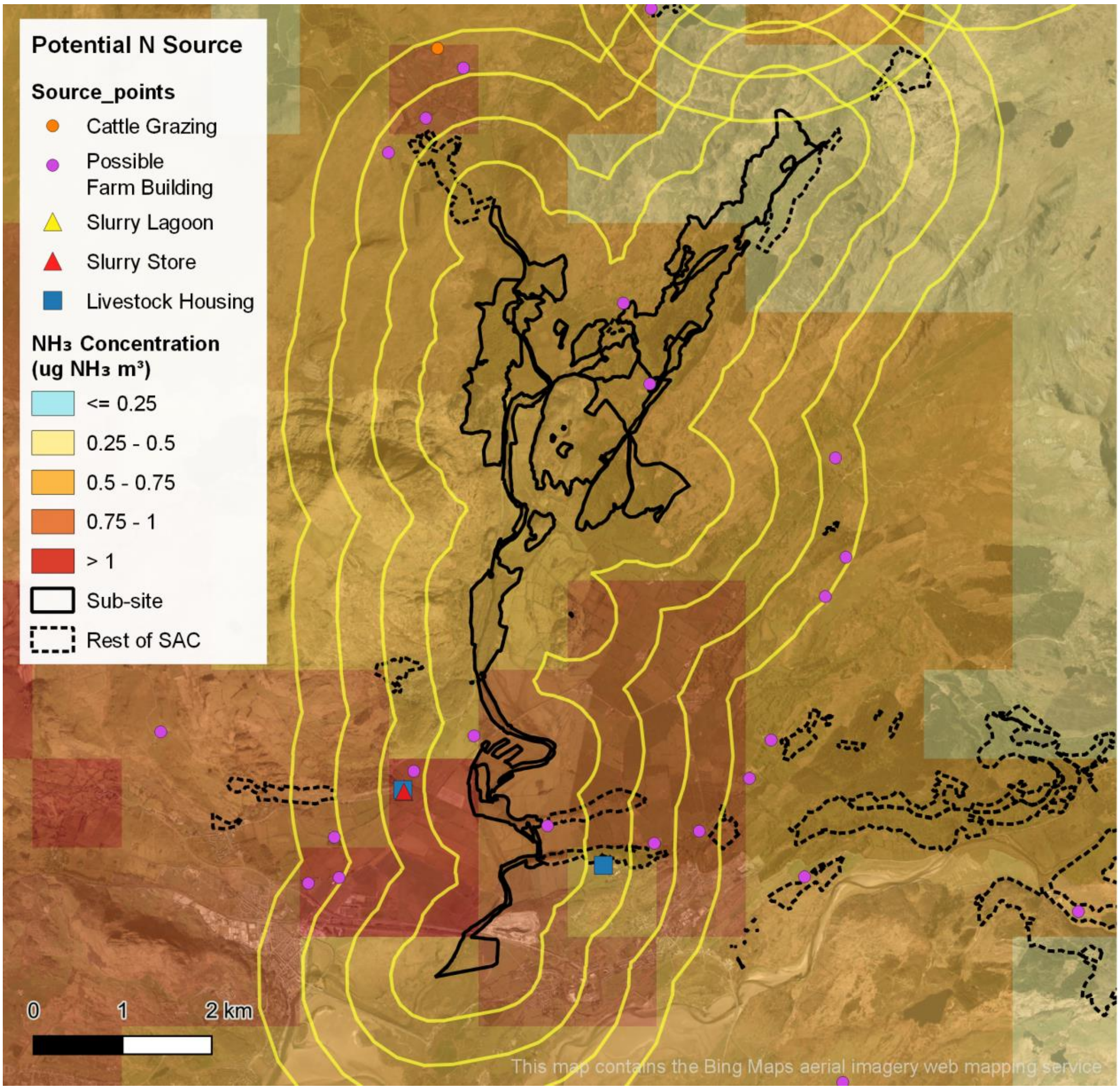

Figure 12: *Hafod Garegog area of Meirionnydd Oakwoods SAC with potential N sources identified from Microsoft Bing Satellite Imagery. 500m buffer zones around sub-site up to 2km shown in yellow.*

Additional assessments of the remaining sub-sites can be found in site reports of Annex A. The Hafod Garegog and Parc Dolmelynllyn sub-sites (discussed above) appear characteristic of the wider Meirionnydd Oakwoods and Bat Sites SAC, with a mixture of lowland and upland agricultural sources across the remaining areas. The area (of Meirionnydd Oakwoods and Bat Sites SAC) surrounding Morfa Harlech National Nature Reserve is situated in a predominantly lowland agricultural landscape and is estimated to have ammonia concentrations above the 1 µg NH_3 m^{-3} critical level. This area appears to have a higher density of grazing cattle than the sub-sites illustrated above, so could be a good candidate for targeted mitigation such as the introduction of livestock exclusion zones and preventing manure or urea fertiliser application in buffer zones directly adjacent to sites. More generally, measures targeted at reducing emissions from livestock can be considered, depending on the current management systems in place. These measures can be associated with the main livestock emission categories as follows:

- housing and outdoor yard areas (e.g. minimising fouled floor areas, appropriate and targeted use of bedding for straw-bedded housing, frequent cleaning of slurry-based housing),
- manure storage (slurry and manure heap covers) and
- slurry and solid manure application (for slurry: use of low emission application machinery, avoiding application under hot, dry conditions, avoiding application in fields bounding the site; for solid manure: similar to slurry; instead of low emission application, rapid ploughing-in).

3.3 Phase 2b: Identification of potential on-site habitat mitigation measures to bring sites into good condition with regard to atmospheric nitrogen

3.3.1 Background of N impacts in woodlands

The effects of N deposition on woodland habitats are reviewed in detail in Bobbink and Hettelingh (2011), and are currently being revised as part of a large-scale revision of critical loads. Empirical critical loads for these habitats are based on changes in the most sensitive components, which include: ground flora, soil processes and nitrate leaching. The draft revision of critical loads for woodlands currently available suggests the lower end of the critical load range remains at 10 kg N ha^{-1} yr^{-1} for deciduous woodland.

Elevated N deposition to woodlands can affect soil processes (e.g. acidification, nutrient cycling and nitrate leaching), tree growth, stoichiometry, and sensitivity to biotic and abiotic stress (Bobbink *et al.* 1996; 1998). Effects on biodiversity occur mainly to three main components: understory vegetation, epiphytic mosses, lichens and alga, and below-ground organisms – primarily macrofungi and mycorrhiza. The most sensitive component of woodlands to N impacts are the epiphytes, leading to changes in epiphyte community composition and loss of sensitive species. With increasing N, there may be increases in tree growth, changes in groundflora or increased growth rates of more nitrophilic groundflora species. Once the soil system becomes N saturated, i.e. there is more plant-available N than the plants and microbes can utilise during part of, or all of, the year, this can lead to adverse effects on tree growth. This manifests through nutrient imbalances, enhanced susceptibility to pests and diseases and to stresses such as drought, ultimately leading to much decreased growth rates and enhanced mortality (Bobbink *et al.* 1996; Etzold et al. 2020). Once N saturation occurs, there is increased risk of nitrate leaching, and soil acidification proceeds more rapidly.

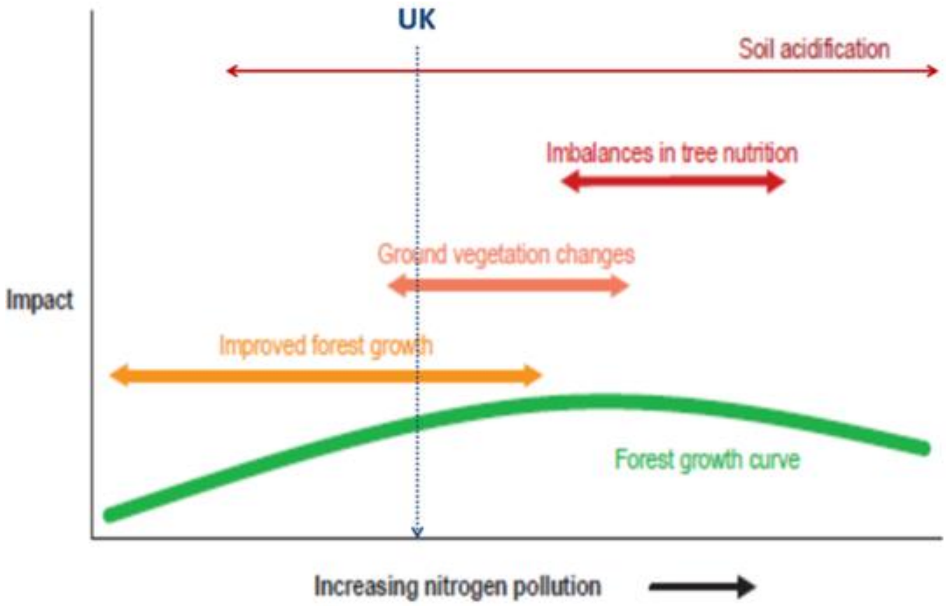

Figure 13: *A schematic representation of the impacts of increased pollution on forest ecosystems (based on Gundersen, 1999). The dotted arrow shows approximately where UK forests currently fit along the nitrogen saturation curve.*

The internal N status of a forest stand reflects historical N deposition and management practice (Emmett, 2002). Soil has a finite capacity to accumulate N, so that as exposure increases retention will decrease and the system will start to 'leak' N. For mature coniferous forests, Gundersen *et al*. (1998) derived three classes of nitrate leaching risk:

- **Low risk:** N limited systems with forest floor C:N>30
- **Moderate risk:** Intermediate systems with forest floor C:N 25-30
- **High risk:** N saturated systems with forest floor C:N<25

In Welsh woodlands, some sites (mainly with Sitka Spruce) have definitively been shown to be N saturated (e.g. Emmett et al. 1993; Wilson & Emmett 1999). Welsh woodlands surveyed in 1998 illustrate some aspects of N saturation (Williams et al. 2000), with substantial leaching fluxes, up to 21 kg N ha^{-1} yr^{-1} primarily as nitrate, at some of the surveyed oakwoods. Leaching fluxes did not directly correlate with atmospheric wet N deposition in bulk precipitation (1992-1994 data from RGAR), i.e. there appeared to be a combination of factors governing leaching, not just high deposition loads. Leaching fluxes > 5 kg N ha^{-1} yr^{-1} were present in oakwoods with bulk deposition ranging from 6.5 to 7.7 kg N ha^{-1} yr^{-1}. Taking into account likely dry deposition, total N deposition to all these woodlands would have been over 10 kg N ha^{-1} yr^{-1} at the time of the study. C:N ratio of the litter layer (O horizon) was a good determinant of leaching fluxes in the Welsh woods, with C:N at sites with high leaching fluxes ranging from 16.2 – 17.9, compared with the majority of other sites with low leaching fluxes having C:N greater than 20 (Williams et al. 2000). They suggested that the risk of N leaching occurred at sites with C:N ratio <20.

Foliar tissue chemistry of both trees and epiphytes can also be a useful indicator of N status, since these organisms accumulate nitrogen in storage compounds in the leaf (sometimes termed luxury N accumulation), when they receive N in excess of their requirements. Foliar chemistry of epiphytes (%N, %P, C:N ratio) is particularly useful as a proxy indicator of N inputs (Rowe et al. 2017). Interpreting the foliar chemistry values is dependent on having a dose-response function since the accumulation threshold and nature of the accumulation curve is species dependent. These curves have been established for a number of moss species and some tree species (Rowe et al. 2017).

In addition to direct N deposition impacts, woodland management plays a large role and may be of greater influence than N deposition, particularly where systems are not N-saturated. For example, was the wood coppiced or high forest, what is the felling and canopy management cycle, is it fenced or not against herbivores. The potential management options in woodlands were reviewed in Stevens et al. (2013), and are summarised & supplemented in the following section.

3.3.2 Woodland management

The Forestry Commission practice guidelines for managing ancient and native woodlands in England (FC, 2010) identify diffuse pollution and inappropriate management as two key threats to these habitats. The definition of ancient and native woodlands is wide-ranging, but can broadly be described as non-plantation woodland. See FC (2010) for a full explanation of the categories which make up this type of habitat. They summarise the ecological benefits of different types of woodland management, but do not address issues of N deposition:

- *Thinning and cutting understorey (Ground cover and shrubs usually < 2 m in height):* releases understorey, enhances ground flora, diversifies species composition, and releases veteran trees.
- *Felling and coppicing:* creates canopy gaps for ground flora and a sheltered woodland edge, and a temporary open phase.
- *Restocking and regenerating:* changes to a more natural mix of species, creates a thicket stage habitat and establishes the next generation of trees.
- *Opening up rides:* enhances woodland edge, restores remnant grassland or heathland habitat, and creates links between bigger patches of open habitat.
- *Managing deer and grazing:* reduces damage to ground flora, allows a shrub layer and understorey structure to develop, and prevents loss of palatable tree species.
- *Conserving deadwood and veteran trees:* conserves micro-habitats that are used by a large proportion of woodland species; remedies an unnatural characteristic of managed woodland, and ensures continuity through the centuries into the future.

3.3.3 Woodland management to mitigate N impacts

Relatively few studies have focused on management to mitigate the impacts of excessive N deposition. Gundersen *et al.* (2006) list five mechanisms that may help alleviate N saturation in temperate forest ecosystems, but other studies cover a range of management options, which are discussed in more detail below. There is also evidence from further afield, primarily the USA (e.g. Clark et al. 2018). The five mechanisms listed by Gunderson et al. (2006) are:

- Reducing N inputs
- Increasing N uptake
- Increasing N export in harvest
- Restoring soil N retention
- Improving catchment-scale N removal in the riparian zone.

These five measures listed above have different impacts on ecological functioning. For example, increasing N uptake in tree biomass or soil N retention may reduce N leaching, but will ultimately contribute to ecological impacts above-ground. The principal on-site measure for woodlands that addresses the most N-related issues is to increase N export in harvest (either of tree material, or litter).

A number of authors (e.g. Fenn *et al.* 2010; Prietzel and Kaiser 2005; Rothe *et al.* 2002) conclude that reductions in N deposition represent the only long-term sustainable method of reducing impacts. However, understanding how site management interacts with N pollution can help develop strategies to mitigate damage in the short- and medium-term. These site management options are discussed below.

3.3.4 Grazing and Browsing

Reported impacts of grazing/browsing in woodlands indicate advantages and disadvantages with respect to offsetting N deposition, affecting tree growth, ground flora and soils. Deer numbers and thus grazing pressures are increasing and need to be managed to protect ground flora in British Woodlands (Kirby 2001). General trends linked to high deer populations include a reduction in *Rubus fruticosus* and all other growing herbs and ferns (other than bracken) and increases in grasses and lower-growing species. Deer browsing in young coppice woodland in eastern England reduced canopy cover and the density and cover of understorey vegetation, while increasing grass cover (Gill and Fuller, 2007). These changes tend to produce similar ecological outcomes as N impacts, and so may not be desirable. Both browsing and shading can reduce understorey vegetation, so one may confound the impact of the other.

Trend data for Wytham Woods (Corney *et al.* 2008) suggest that changes in soil pH and nutrient status and deer browsing combined can change species richness and composition leading to an increase in grass species. Deer preferentially browsed *Rubus fruticosus* in shaded (i.e. closed canopy) areas helping to alter the shrub layer and competitive interactions between species leading to an increase in grass species.

Deer browsing also impacts on soils. Litter decomposition rates in native regenerating birch woodland in the Highlands of Scotland were significantly reduced by deer browsing (Harrison and Bardgett 2003). Browsing reduced litter quality, suggesting that herbivores can reduce rates of nutrient cycling in this habitat, and potentially mitigating some adverse effects of N deposition impacts on ground flora. In a regenerating woodland in northern Britain growth of *Betula pubescens* was N limited in browsed areas reflecting lower N mineralisation rates (- 50%) and lower N availability. Tree growth rates and the quantity and quality of litter returned to the soil were all lower in browsed areas compared with unbrowsed areas. Both these studies suggest that deer browsing can help restrict the impacts of N deposition on soil N status. On the downside however, browsing itself can cause similar, deleterious effects on the understorey vegetation to excess N deposition.

3.3.5 Litter removal

Litter removal can reduce the amount of N in the ecosystem, but it also depletes the soil of other important nutrients, so the amount removed would need to be optimized. Litter removal could also have significant adverse impacts on the invertebrate fauna (e.g. leafmining lepidoptera, gall wasps etc), and is unlikely to be an acceptable option for the majority of protected areas. However, it is reviewed here for completeness. A litter removal experiment over a 16 year period, in an acidophilous mixed oak-pine woodland in southern Poland (Dzwonko and Gawronski, 2002), resulted in substantial impoverishment of the soil, with plots containing significantly less P, Mg, Ca and lower cation exchange capacity (CEC). This would very likely contribute to soil acidification, but in principle is little different from removal of N and other elements during harvesting. Vascular plants and bryophytes colonised these plots much more frequently, increasing overall species richness. In the control plots (no litter removal) vegetation changed from acidophilous to neutrophilous, and vascular plants and mosses disappeared, due to the thick litter layer impeding seed germination and development, and competition by dominant species.

Six years of intensive prescribed litter raking in N saturated Scots pine forest in southern Germany reduced the soil N pool by 450 kg ha^{-1} (equivalent to 75 kg N ha^{-1} yr^{-1}) (Prietzel and Kaiser 2005). The removed N was approximately 11% of the original pools down to 100 cm soil depth, and forest floor N pools were reduced by 40%. The bolewood was estimated to sequester only 22 kg N ha^{-1}. Therefore, prescribed litter-raking could provide an effective tool to support endangered ground vegetation species adapted to N limitation and frequent ecosystem disturbance; and keep nitrate concentrations in seepage water low.

3.3.6 Thinning or harvesting

Although the focus of this report is on non-productive (i.e. non-commercial) forest, it may be useful to note that estimates of N removal by harvesting in productive UK forests, are in the order of 2.9 kg N ha^{-1} yr^{-1} for coniferous woodland and 5.9 kg N ha^{-1} yr^{-1} for broadleaved woodland (Hall *et al.* 2003).

A meta-analysis of forest management and soil C (Johnson and Curtis 2001) showed that on average, forest harvesting had little or no effect on soil C or N. However, significant effects of harvest type and tree type, deciduous versus evergreen, were identified; sawlog harvesting increased soil C and N (+18%), contrasting with a 6% decrease from whole-tree harvesting. This positive effect of sawlog harvesting appeared to be restricted to coniferous species. Fertilisation and naturally invading vegetation associated with N fixation increased soil C and N overall. In an ancient deciduous woodland in Cumbria, the characteristic ground flora changed considerably over 18 years at sites either left unmanaged, or cleared and replanted, becoming clearly distinct from that of a traditionally managed wood through practices such as coppicing (Barkham 1992).

Thinning will alter the forest canopy and thus the amount of light accessible to ground flora. In 20 beech woodlands across the UK vegetation cover tended to be greater (>50%) where the canopy gap fractions exceeded 9% (Kennedy and Pitman 2004). The ground flora response to thinning may take time: two years was not sufficient to show changes, whereas after five years gaps were dominated by swards of *Holcus* and *Agrostis* grass species, which may not be desirable. The study also showed that the effects of light on ground flora can vary according to the age, structure and management of woodlands. At some mature tree sites the age of the trees had resulted in lower branch dieback and openings in the canopy permitting mosses to colonise the forest floor at some sites, or brambles (*Rubus* spp.). Although brambles can suggest N enrichment, Kennedy and Pitman (2004) found no significant relationship between incoming N deposition (estimated from national deposition databases) and ground flora composition. There was however, a relationship between weighted Ellenberg N score mean site values and the average distance to the edge of the woodland, indicating the importance of local N sources in determining ground flora species composition.

Hardtle *et al.* (2003) also examined the effects of light and soil conditions on the species richness of ground vegetation in three types of deciduous forest in northern Germany. In moist forests of alder-ash, species richness of the ground vegetation was positively correlated with soil moisture, while light and nutrient supply appeared to have no effect. In meso- to eutrophic beech forests, where many ground cover species are shade tolerant, species richness was determined by nutritional status (closely correlated with soil activity and base and N supply) rather than light. By contrast, in acidophyte beech and mixed beech-oak forests, species richness did respond to canopy closure and interior light conditions. This study also highlights how effects of soil moisture, nutrient supply and light conditions on ground flora depend on the type of forest community. It should also be appreciated that N as a nutrient is key to light harvesting, being a fundamental component of the Rubisco enzyme. This means that increasing light levels can trigger similar responses in woodland ground level to N eutrophication, encouraging graminoids at the expense of lower plants and herbs. However, N deposition impacts on groundflora can depend on the type of ground flora present, as well as factors such as light levels and soil fertility (Walter et al. 2016; Walter et al. 2017; Perring et al. 2018).

3.3.7 Burning

Prescribed burning is used as a forest management tool in some countries, including the USA. Fire can reduce ecosystem total C and N pools, and open up habitat areas. However, fire is not a naturally occurring phenomenon in UK woodlands, and prescribed fire is not a suitable management option for removing N from UK woodland ecosystems, since they contain sensitive ground vegetation and epiphytic lichen communities which would be adversely impacted by fire.

3.3.8 Evidence from further afield

Clark et al. 2019 conducted a meta-analysis of the effectiveness of four remediation approaches, focused on North American forests. The approaches were: prescribed burning, thinning, liming, carbon addition. The assessment of responses focused on a range of metrics for recovery from N deposition (decreased soil N availability, increased soil alkalinity, increased plant biodiversity).

Approaches that reduce soil N availability work via a range of mechanisms. Enhancing soil microbial immobilization can be achieved through carbon addition or thinning with slash left onsite (Clark et al. 2010), but these techniques tend to add to or stabilise existing N in the system. Direct removal of soil N pools can be achieved by prescribed burning, litter/topsoil removal, planting and harvesting (Boerner et al. 2008, Jones et al. 2017, Boxman and Roelofs 2006). These have the effect of removing accumulated soil N stocks. Direct removal of aboveground N is perhaps the easiest technique, through prescribed burning, mowing, grazing, planting and harvesting, thinning with slash removal (Boerner et al. 2008, Boxman and Roelofs 2006, Jones et al. 2017). This may not always be possible in semi-natural/non-commercial woodlands.

Other approaches counteract some of the effects of N deposition, e.g. counteracting soil acidity through liming (Likens et al. 1996, Driscoll et al. 2001, Lawrence et al. 2016, Błońska et al., 2015; Huang et al., 2014).

Approaches can also make habitats more suitable by increasing light levels at the soil surface and opening up sites for colonization, through prescribed burning, thinning, mowing, grazing (Jones et al. 2017). Lastly, for restoring sites impacted by N deposition, propagules can be re-introduced or exposed that may either be dormant in a deep seed bank or locally extirpated. This can be done via litter/topsoil removal, or replanting or seed scattering (Bakker and Berendse 1999). However, it does not address the underlying problems of excess N input, and excess N in the woodland system, which led to the problems in the first place. Without addressing these issues, any attempt at restoration of individual species is likely to fail.

Carbon addition was the only treatment that decreased N availability (effect size: -1.6 to -1.8), while liming, thinning, and burning all tended to increase N availability (effect sizes: +0.2 to +1.1). Only liming had a significant effect on soil alkalinity (+10% to 80% across metrics). Only prescribed burning and thinning affected plant diversity, but with opposing effects across metrics (i.e. increased richness, decreased Shannon or Simpson diversity) and effects were often statistically marginal.

From this study, it appears that no single treatment will be effective in promoting recovery from N deposition for all three responses of interest, and combinations of treatments should be explored, depending on the desired outcome.

The approaches described above are geared towards soil-mediated impacts, primarily on ground flora, and are not directly beneficial to epiphytes, and in many cases may be damaging to epiphytes (e.g. fire, grazing).

3.3.9 Recommendations

The following table summarises recent guidance on management for woodlands to reduce N impacts (adapted from Stevens et al. 2013).

Table 9: *Summary table of management recommendations, which could be used to reduce nitrogen impacts or speed recovery for woodland habitat. A confidence of 1 means that there is strong evidence to support the recommendation, 2 means there is some evidence to support the recommendation and 3 means more evidence needs to be collected.*

Recommendation	Justification	Confidence
Do not apply fertiliser	Avoids adding N to the site	1
Do not supplementary feed livestock	Avoids adding N to the site	1
Continue to retain deadwood	N remains on site but in low quantities. Conservation benefits of deadwood outweigh damage.	2
Avoid ring barking, except where used to control invasive species	Avoids N returns to soil	3

In addition, it may be worth considering litter removal in some instances in upland situations, which can remove N stocks and has the potential to reduce some N deposition impacts. However, it may not be suitable in some sites due to steep topography. There has been very little investigation into the impact of this practice on N cycling and unintended consequences on ground flora, litter and soil fauna, but further work should be undertaken to investigate its potential, and to assess the costs of revising this technique.

Increasing grazing/browsing can have both positive and negative impacts with respect to N cycling and N availability. At present it is difficult to recommend a particular view on this from an N recovery perspective.

As noted above, the majority of these management strategies are focused on trees and ground flora, and may not be suitable or appropriate for epiphytes. There do not appear to be many on-site measures which can alleviate N impacts on epiphytes.

3.4 Phase 3 – Guidance on implementing SNAPs and an approach for auditing effects

3.4.1 Guidance on implementing SNAPs

National scale modelling of nitrogen emissions, concentrations and deposition are an important resource for understanding potential atmospheric nitrogen threats to designated sites (as demonstrated in Phase 1). The national datasets used in Phase 1 can help to prioritise sites in which to propose Site Nitrogen Action Plans (SNAPs) and where to spatially target off-site mitigation efforts. It is however vital that these national datasets are combined with local knowledge and information, as emission activities surrounding sites can be highly variable and concentration/deposition hotspots may be overlooked due to averaging e.g. over a grid-resolution of 1km x 1km (or 5km x 5km grid resolution), or across sites. It is therefore important to identify any significant emission sources surrounding sites (especially those close to the site boundary or on the site itself). Initially this can be done remotely, using aerial and satellite imagery (as demonstrated in Phase 2a), but visiting the site/contacting site officers may reveal further details. Once a site or sub-site has been selected to implement a SNAP, it is important to consider the following steps before implementation/intervention:

- **Communications with local stakeholders:** Engagement with local stakeholders is key to the SNAP process for several reasons: a) for creating a common understanding of the threats, building trust and agreeing the need for action, and b) for helping in the assessment of current and likely future threats to the site from atmospheric N input and recommendation for possible on-/off-site measures to mitigate effects (in parallel). Engagement could be made through facilitated meetings or by contacting trusted local persons such as Catchment Sensitive Farming Officers directly.
- **Ecological assessments:** Gaining an understanding of the current ecological and nitrogen status of a site (inc. woodland, epiphytes, ground flora and soil) is essential for effects (and responses to any on/off site measures) to be accurately audited. This should include (where relevant) soil type, litter C:N ratio, plant foliar N. Modelled deposition history datasets (e.g. cumulative N and S deposition over the last 30 years) could also be examined to identify potential legacy effects.
- **Atmospheric monitoring:** Monitoring atmospheric NH_3 concentrations and wet N deposition prior to intervention is essential for setting a baseline against which the success of any mitigation efforts can be assessed, and to more accurately measure the current level of pressure on the site. Monitoring should be carried out for at least one year prior to any implementation of measures, so that any seasonal trends in activity are captured prior to intervention.
- **Identifying funding mechanisms:** An agreement on the way forward and clarification on any funding that could be made available for implementation of any on-/off-site measures. For example this could be supported under the Sustainable Farming Scheme (SFS) for Wales.
- **Implementation of measures**: Once initial stakeholder communications and any pre-intervention measurements and modelling have been completed and suitable measures for the site have been agreed, the measures are implemented. The AAANIS report (Carnell and Dragosits 2015) suggested taht these local measures could be implemented under a SNAP DRAGONS (Delivering Regional Action on the Ground On Nature Sites) framework, to minimise effects from local sources. Monitoring of atmospheric N (monthly) and some ecological assessments (at appropriate intervals, depending on the indicator) are carried out throughout the process, and analysed for any change occurring,
- **Auditing of recovery** (see 3.4.2. for details) Some indicators (e.g. NH_3 concentrations near sources will change faster than the ecological indicators upon implementation of measures, with recovery processes depending on habitat types, magnitude of change, level of damage etc. Therefore, auditing will have to be carried out at the appropriate time scale for the site and the ecological indicators, according to the measures implemented.

3.4.2 An approach for auditing effects of on- and off-site measures with SNAPs

In order to better understand the potential current, or any future, N impacts on any site, we recommend the following monitoring to be established (before the implementation of any measures, to create a baseline for current and historical levels of pressure and impact, as described in Section 3.4.1):

- **Measure atmospheric nitrogen input to the site**: Measured data are more accurate than modelled data (which at best give 1 x 1 km grid average information, based on emission inventories and models which have their own uncertainties). This is especially relevant near to point sources, where steep gradients may occur at sub-1km grid resolution – these would be hidden in average modelling data. The following components should be assessed:
    - **Atmospheric NH_3 concentrations**: The most cost-effective and flexible approach is to use passive samplers which do not need electricity (monthly sampler exchange, analysis in the lab). These can be set up across the site to establish concentration gradients across the site and identify any areas with higher risk. The spatial patterns identified will be useful for informing decisions on where (if any) spatially targeted measures may be needed to mitigate local emission hot spots. The type of sampler used should take into account the concentrations to be monitored, and should have appropriate limits of detection.
    - **Atmospheric N deposition** to the site: The two main components, wet and dry deposition, can be quantified as follows:
        - **Wet deposition**: direct measurement by collecting water samples twice monthly and analysing in the lab. Wet deposition is generally much less spatially variable in relation to emission sources, and rainfall patterns etc. are more important to determine N input to a site from this source.
        - **dry gaseous NH_3 deposition:** derived from monitoring of NH_3 concentrations.
- **Soil chemistry:** particularly the pH and C:N ratio of the litter layer (O horizon). This reflects the site's pollution deposition history and gives a good indication of N saturation.
- **Plant foliar chemistry:** for woodland habitats, samples of epiphytic lichens and mosses can be collected and analysed in the lab, and foliar chemistry of tree and understory vegetation can also be carried out. We suggest to measure %N, %P and %C. This can indicate levels of N deposition experienced within the last few years from atmospheric sources.
- **Species inventories of ground flora** (using permanently marked quadrats which can be revisited in future), **and epiphytes** as well as other species of potential interest, including fungi. These allow assessment of the current status of the plant community, which can be bench-marked against monitoring data from other sites, as well as acting as a baseline for assessing recovery (or potentially worsening of impacts) in the future.

4 Discussion & conclusions

The five selected study sites are relatively clean overall, with ammonia concentrations typically below 1 µg NH_3 m^{-3} and therefore below the most sensitive critical level for lichens, mosses and bryophytes. Agricultural emission density estimates are also relatively low in areas surrounding sites, with high proportions of emissions from sheep (indicative of more extensive agriculture). However, although NH_3 emissions and concentrations surrounding sites are relatively low (in comparison to the more intensively farmed parts of Wales and the wider the UK), the sites are estimated to be in exceedance of the critical loads (related to atmospheric N deposition) for the sites' designated features. N deposition is often from further afield (regional to wider national and international sources), and therefore fewer local interventions are typically available, and more substantial and wide ranging efforts may be needed to decrease these long-range components of N deposition. Proposed emission reductions from agricultural sources across Wales under the SFS for example, will likely benefit these sites.

The study shows the importance of detailed local mapping of emissions sources, and of sensitive features within sites. Although a substantial proportion of deposition onto a given site arrives by long-range transport, nearby ammonia sources can have a dramatic impact on sensitive features. The concentration of ammonia declines steeply over the first 200 m away from an intensive source, but the influence of larger sources can be seen for greater distances. Damage through excess atmospheric N could be considerably reduced by ensuring that sources are separated from particularly sensitive features by sufficient distance and/or the presence of buffer woodland (which can help to disperse, dilute and recapture a proportion of the atmospheric ammonia, if the woodland is optimally designed and located in relation to emission sources and sinks..

The coordinates of farm (holding) locations provided by Welsh Government enabled more accurate source mapping than is possible from aggregated parish-level data which are used for the agricultural ammonia emission inventory (to avoid data disclosure which is a requirement for use of these statistics). Therefore, the need for data protection in relation to individual holdings puts a limit on the accuracy of source mapping. More accurate data on the locations of slurry stores etc. could be used, e.g. by local surveying, but this would require specific data collection and mapping for each site. It should be noted that the calculated emissions are based on average practice, which is in turn based on land use, herd size etc. Therefore, the modelled emission data do not necessarily reflect current practice, but provide a best estimate based on the available information. Where mitigation measures may have been implemented at a farm, these may not necessarily be reflected in modelled estimates of emissions, as only average national figures are available rather than per-farm data. The concentration and deposition model output is based on these average practice emission, thereby contributing to the uncertainty of the atmospheric N affecting the site. Better data availability (and permission to use the available data at a per-holding level could substantially improve the spatial modelling and remove the need to anonymise source categories in the emission density dataset (where fewer than 5 holding contribute to a category). Detailed, up-to-date maps of emissions sources would allow the effects of mitigation measures to be assessed, and good practice by land managers to be acknowledged and potentially rewarded.

Detailed mapping of land use beyond the site boundary is also useful for assessing potential N impacts. As well as mapping land uses that are sources of ammonia (i.e. improved grass or arable land), information on the location and extent of nearby woodland is useful for estimating the recapture of ammonia and consequent effects on NH_3 concentration and nitrogen deposition within protected sites.

Better information on the distribution of sensitive features within sites would also improve the protection of biodiversity and the cost-effectiveness of measures. Detailed site maps enable a more confident assessment of risk. Much information is often available in local records, and it would be valuable to coordinate SNCA and UKCEH initiatives to digitise and maintain maps and records for protected sites and for priority habitats. For habitat mapping by the UK National Focal Centre, some habitats are distinguished using records of plant and lichen species occurrence. There may be potential to use botanical records to identify the locations of rare habitats and species and so improve the targeting of NH_3 emissions control measures.

The damaging effects of atmospheric N pollution are increasingly recognised by site managers and farmers. The approaches to modelling and data visualisation used in the current study could be applied more widely to inform discussions of local permitting and targeted measures. Such targeting could improve the protection of key habitat locations and species, if local data are available to support the implementation of measures. Local analysis and mapping need to be reasonably accurate if they are to support decisions about agri-environment support and legislation.

The approach can be summarised the SNAP DRAGONS (Site Nitrogen Action Plans Delivering Regional Action on the Ground On Nature Sites) framework. This consists of the development of a SNAP (Site Nitrogen Action Plan) followed by the implementation of measures and monitoring (DRAGONS) split into 3 phases:

1) **Analysis of available data:** This includes a) national scale modelled data to identify key threats from atmospheric N input to sensitive protected sites, in particular to their designated features of interest (in this study, woodland features across 5 SACs), b) publicly available aerial photo mapping, and c) any relevant atmospheric N measurement data, such as from the UK national monitoring networks.
2) **Identification of suitable mitigation measures:** This includes a) off-site measures (e.g. agricultural mitigation measures such as low-emission slurry spreading, covering of slurry stores) and b) on-site habitat mitigation and restoration measures.

Information gathered during Phase 1 and Phase 2 needs to be refined and re-evaluated under Phase 3, if needed.

3) **Stakeholder engagement, implementation and auditing of the SNAP:** This includes a) engagement and building of trust with local stakeholders, b) further evidence gathering on local emission sources (e.g. mitigation already in place, technical assessment of options), c) establishment of baseline measurements of atmospheric N and habitat condition/indicators to assess current levels of impact (or absence of impact) and act as a baseline against which future change can be assessed, d) confirming of funding mechanisms for measures and implementation, e) monitoring/auditing of atmospheric N input and recovery.

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