Defra Contract AQ0826:

Modelling and mapping of exceedance of critical loads and critical levels for acidification and eutrophication in the UK 2013-2016

Final Report: 25th July 2016

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EXECUTIVE SUMMARY

This report covers the work of the original contract (2013-15) and the following one-year extension (2015-16). The overall purpose of this project was to maintain, and where appropriate update, the UK critical loads database, and to provide estimates of critical load and critical level exceedance based on current pollutant deposition or concentrations, and scenarios for the future. The exceedance results were used to inform policy makers on the areas of sensitive habitats and designated sites potentially at risk from air pollution and were updated annually to provide a UK indicator of the impacts of air pollution on ecosystems. The project also supported the UK National Focal Centre (NFC) for critical loads modelling and mapping. The 1-year extension to this contract additionally included the biodiversity modelling required to enable the UK NFC to begin work in preparation for responding to the 2015-17 "Call for Data" under the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP). The key results and recommendations are listed below.

<u>UK critical load databases</u>

- No updates were made to the national 1km habitats critical loads database during this contract.
- The Site Relevant Critical Load (SRCL) tables linking SAC, SPA and SSSI designated features to European Nature Information System (EUNIS) habitat classes were updated in 2014 in collaboration with JNCC.
- SRCL values for nutrient nitrogen were updated in 2014 to the values recommended by the Statutory Nature Conservation Bodies (SNCBs) for casework and EU Habitats Directive Article 17 reporting.

<u>UK concentration and deposition data</u>

- Concentration based estimated deposition (CBED) 5km data sets from 2004-06 onwards were updated in 2015 to correct for an over-estimation of nitric acid deposition.
- Long-term trends (1995 to 2014) in deposition budgets, based on CBED data to moorland and woodland, showed a ~64% reduction in non-marine sulphur and a 15-20% reduction in total (oxidised plus reduced) nitrogen.
- For calculating national-scale exceedances of critical levels of ammonia we recommended using FRAME 1km ammonia concentration data as this improved the spatial separation of ammonia source and sink areas.

<u>Exceedances of critical loads and critical levels</u>

- The long-term trends (1995 to 2014) in critical load exceedance showed a 28.5% reduction in the area of UK acid-sensitive habitats with exceedance of acidity critical loads and a reduction of 64% in the Average Accumulated Exceedance (AAE). For nutrient nitrogen the reductions were smaller, reflecting the smaller reductions in nitrogen deposition compared to acid deposition, with a 12.8% reduction in the UK area of nitrogen-sensitive habitats with critical load exceedance and a 36.7% reduction in AAE.
- The latest national habitat critical load exceedance results based on CBED deposition data for 2012-14 showed 44.1% of the UK area of acid-sensitive habitats have exceedance of acidity critical loads, and an AAE of 0.28 keq ha^{-1} $year^{-1}$. For nutrient nitrogen, critical loads were exceeded across 62.2% of the UK area of nitrogen-sensitive habitats, with an AAE of 6.0 kg N ha^{-1} $year^{-1}$.

- The long-term trends (1995 to 2014) in exceedances of SRCL for UK SACs, SPAs and SSSIs showed a reduction in the percentage of sites with at least one feature exceeded, of 15-23% for acidity and 5-8% for nutrient nitrogen.
- The latest UK SRCL exceedance results based on CBED deposition for 2012-14 showed that 76% of SACs, 70.3% of SPAs and 61.4% of SSSIs had exceedance of acidity critical loads for at least one feature in a site. Nutrient nitrogen critical loads were exceeded for at least one feature in 90.1% of SACs, 73.3% of SPAs and 88.1% of SSSIs.
- Ammonia concentrations (FRAME 1km data) for 2011-13 exceeded the critical level of 1µg m^{-3} across 62.7% of the UK land area; 3.8% of the UK land area received concentrations above the critical level of 3 µg m^{-3}. The percentage of UK SACs with ammonia concentrations above the critical levels anywhere across a site, was 60.9% for the critical level of 1µg m^{-3} and 7.5% for the critical level of 3 µg m^{-3}.

<u>Inputs to the Defra Model Intercomparison Exercise</u>

- Deposition data sets from different models (CBED, FRAME, EMEP4UK, NAME, CMAQ) were processed to convert data to the same grid system. These data, together with the calculated deposition budgets for the UK were provided to David Carslaw to complete work on the Model Inter-Comparison project (Defra contract AQ0936).
- A short report was submitted to Defra in July 2015 on the results of a preliminary comparison of ecosystem-specific deposition from CBED and FRAME and the associated critical load exceedance results. Further work would be required to include EMEP4UK in this comparison.

<u>Project websites</u>

- Two new project websites were created:
    - National critical loads and dynamic modelling: http://www.cldm.ceh.ac.uk
      This was created to provide information on the role of the National Focal Centre; critical load definitions, their calculations and use; exceedance metrics, maps and trends; publications and downloadable reports, including the "Methods Report" and "Trends Report".
    - Pollutant deposition: http://www.pollutantdeposition.ceh.ac.uk
      This was created to provide downloadable deposition datasets and maps and information about the pollutant monitoring networks.

<u>Activities of the UK National Focal Centre for critical loads modelling and mapping</u>

- The project provided UK representation at annual meetings of UNECE Task Force of the International Cooperative Programme on Modelling and Mapping (ICP M&M) and workshops of the Coordination Centre for Effects (CCE).
- The NFC submitted the following UK data in response to calls from the UNECE Working Group on Effects (WGE) and CCE:
    - 2012-14 data call: Biodiversity metric (mean rescaled habitat suitability for locally-occurring positive indicator species) values for 18 sites, representing 9 different EUNIS habitat classes and two deposition scenarios.
    - 2014-15 data call: (i) Habitat suitability data for 40 sites (26 acid grassland and 14 heathland); (ii) National 1km acidity and nutrient nitrogen critical loads for UK habitats sensitive to acidification and/or eutrophication; (iii) Nutrient nitrogen critical loads for designated features of SACs and SPAs.

- The MADOC-MultiMOVE-HQI model chain has been developed and applied to the acid-sensitive habitats (bogs, dry acid grasslands, wet heaths, dry heaths) of 354 SACs to demonstrate the methodology for deriving new biodiversity-based critical load functions in the UK that fully take into account the combined effects of sulphur and nitrogen on both acidification and eutrophication and the associated impacts on biodiversity. The overall form of most of these functions was satisfactory, in that they showed increases in overall habitat quality with decreases in both nitrogen and sulphur deposition. Preliminary results were presented at the joint meeting of the ICP M&M and CCE in April 2016.

Project reports

- The key reports produced under this contract were:
    - A "Methods Report" on the calculation and mapping of critical loads and their exceedances in the UK, including site relevant critical loads.
    - A "Trends Report" (updated in May 2016) providing a summary of the calculation of exceedances and presenting trends in (i) critical loads for UK habitats sensitive to acidification and/or eutrophication; (ii) exceedances of SRCL for UK SACs, SPAs and SSSIs.

  Both reports are/will be available to download from the project website: http://www.cldm.ceh.ac.uk/

Analysis of FRAME deposition scenarios

- FRAME deposition data sets were generated for 2025 and 2030 based on the UEP45 emissions scenario. Critical load exceedances for UK habitats sensitive to acidification and/or eutrophication were calculated for both scenarios, and both gave very similar results with less than 1% difference in the UK habitat area exceeded, and virtually the same AAE. The 2025 scenario showed a 6% reduction in the UK area of acid-sensitive habitats with exceedance of acidity critical loads, and a 19% reduction in the AAE, compared with the present day (CBED 2010-12 data, prior to 2015 update). For nutrient nitrogen the reductions were very small: 0.9% reduction in the UK area of nitrogen-sensitive habitats exceeded and a 4% reduction in the AAE.
- FRAME deposition scenarios were generated based on 6% and 10% reductions in sulphur deposition from shipping and compared with a baseline scenario. Differences in the critical load exceedance results for all three scenarios were small; the 10% reduction in sulphur deposition reduced the UK area of acid-sensitive habitats exceeded by 0.6% (460 km^2) compared to the baseline.

A new method for calculating acidity critical loads for peat soils

- A new method was developed for calculating acidity critical loads for peat soils, based on the buffering of acidic inputs that may be provided through the reduction and subsequent incorporation of sulphate into accumulating peat organic matter. This new method is currently only applicable to upland bog habitats in good condition; for these habitats it resulted in higher critical loads and smaller or no exceedances.
- Peatlands that have been degraded (including those in high deposition areas of the UK) by land management activities such as burning or drainage have a reduced capacity to retain pollutants, and are therefore more vulnerable to acidification. Further work is required to develop and apply the method to degraded bog/peatland habitats, and national-scale maps of bog condition would be required to apply the new method to all bog/peatland habitat types in the UK.
- The new method assumes there is no net effect of nitrogen deposition on the acid neutralising capacity of peatlands and therefore only considers the acidification risk from sulphur deposition.

However, this does not mean that bog/peatland habitats are not at risk from the impacts of nitrogen deposition; the empirical critical loads of nutrient nitrogen should still be applied to bog habitats in the UK.

<u>Provision of data and advice</u>

- The project provided advice and critical loads and deposition data to Defra, JNCC, Scottish Government, Welsh government, SEPA, Natural England, and Ricardo-AEA.
- The project carried out an ad-hoc study for Defra and JNCC to illustrate the ammonia deposition reductions that may be required in the future to meet protection targets for Annex I habitats.
- Members of the project team attended and gave presentations at annual meetings of the Committee on Air Pollution Effects Research (CAPER); a forum for informing policymakers and scientists on developments in the assessment of air pollution impacts on the natural and semi-natural environment.

1. Work Package 1: Maintenance of UK critical loads database

Summary

- *No updates have been made to the national 1km habitats critical loads database during this contract.*
- *The Site Relevant Critical Load (SRCL) tables linking designated features to EUNIS habitat classes were updated in 2014 in collaboration with JNCC.*
- *SRCL values for nutrient nitrogen were updated in 2014 to the values recommended by the Statutory Nature Conservation Bodies (SNCBs) for casework and EU Habitats Directive Article 17 reporting.*

A new database structure was set up for this project to pull together the critical load databases, deposition data sets and other related data from the previous contract(s), and provided a structure appropriate for the data analysis requirements of this project. The two main critical loads databases maintained and used under this project are described below.

1.1 UK Habitats sensitive to acidification and/or eutrophication

The critical loads for terrestrial habitats are mapped on a 1km grid and the data stored in both Access database tables and as ArcGIS 1km gridded maps. The data include the following:

- Acidity critical loads for soils (CLA)
- Acidity critical loads for 1752 freshwaters (CLA, CLmaxS, CLminN, CLmaxN)
- Acidity critical loads (CLA, CLmaxS, CLminN, CLmaxN) for eight terrestrial habitats
- Nutrient nitrogen critical loads for 13 habitats (CLnutN)

Where: CLA = acidity critical load

CLmaxS = acidity critical load expressed in terms of sulphur only (ie, when nitrogen deposition is zero)

CLmaxN = acidity critical load expressed in terms of nitrogen only (ie, when sulphur deposition is zero)

CLminN = amount of nitrogen removed through immobilisation, denitrification and harvesting by vegetation

CLnutN = critical loads for nutrient nitrogen (eutrophication)

The acidity critical loads CLmaxS, CLminN and CLmaxN are required for the calculation of exceedance by sulphur and nitrogen (ie, acid) deposition. In general, mass balance methods are used to calculate critical loads for managed (productive) woodland in the UK, and empirical methods for all other habitats. Further details on calculating critical loads and a description of the methods used to map the habitat distributions can be found in the "Methods Report" (Hall et al, 2015a). The national 1km critical load maps only include data for the habitat areas included in these habitat distribution maps; the critical loads database includes the area of each habitat mapped in each 1km grid square of the UK and these were used to assess the areas of ecosystems at risk (WP3).

A new method has been developed for calculating acidity critical loads for peat soils (WP10). Further work is required before this new method could be applied to all bog or peatland habitats in the UK; if this method is approved in the future, the acidity critical loads for habitats on peat soils (eg, bog, and potentially some areas of wet heath or wet acid grassland) will need to be updated.

To date, nutrient nitrogen critical loads have only been applied to terrestrial habitats; they have not been set nationally for UK freshwaters due in part to a lack of sufficient UK data on which to base the critical loads. However, the evidence base for nutrient nitrogen impacts on freshwaters is developing; Appendix 1 provides a list of some relevant papers.

The national critical loads database has been made freely available to users on request under a CEH data licence agreement. The national critical loads database and the SRCL database (see below) have been used for UK Integrated Assessment Modelling activities under the Defra SNAPS project (AQ0947), and are also used in the Air Pollution Information System (APIS: www.apis.ac.uk).

1.2 Site-Relevant Critical Loads (SRCL) for SACs, SPAs and SSSIs

These are the critical loads that have been assigned to the habitat features of designated sites (Hall et al, 2015a) to enable assessments of the number and area of sites at risk from the adverse impacts of excess acid or nitrogen deposition. Digital information on the spatial location of feature habitats is not currently available; therefore in UK assessments based on these data, each feature habitat is assumed to occur across the entire site. Nutrient nitrogen critical loads are based on the empirical values agreed at national (Hall et al, 2011a, 2015a) and international workshops (Bobbink & Hettelingh, 2011). They were assigned by using look-up tables that relate the feature habitats to the EUNIS habitat classes (Davies & Moss, 2002) used by Bobbink & Hettelingh (2011). For acidity, an additional critical loads database was used that is based on the same habitats and methods as the database outlined in Section 1.1, but provides critical load values (CLA, CLmaxS, CLminN, CLmaxN) for all habitats for all 1km squares of the UK (ie, the data are not masked to the habitat distributions). This enabled acidity critical load values to be extracted and assigned to sensitive habitat features, even where those features may cover a very small area and not appear in the national-scale habitat distribution maps used (Section 1.1). The SRCL data are stored in an Access database and linked spatially to the SAC, SPA, SSSI boundaries held in ArcGIS.

During 2013-14 JNCC and Natural England reviewed and updated the linkages between site features and EUNIS habitat classes and the associated nutrient nitrogen critical load values recommended for use in Article 17 reporting under the Habitats Directive. For some habitats this “recommended” value was the same as the UK “mapping value” (ie, the critical load assigned to the 1km habitat squares of the data in Section 1.1 above) based on UK evidence of nitrogen impacts; for other habitats, or where there is no mapping value available, the minimum of the published critical load range was used (Table 1.1).

Currently the feature habitats of SSSIs are recorded in the UK SRCL database by broad habitat type. Natural England have changed (or will be changing) to using the species communities of the National Vegetation Classification (NVC; Rodwell, 1991-2000) to record their SSSI designated features. If the UK SRCL database component for England is updated in the future using NVC classes, it will require new programs to be written for the calculation of exceedances using the NVC categories, and to re-integrate the analysis with that for the rest of the UK that is using broad habitat categories.

If the new method for calculating acidity critical loads for peat soils (WP10) is further developed and approved for application to all bog or peatland habitats, the SRCL database will need to be updated in the future for sites containing these habitat types.

Table 1.1 Nutrient nitrogen critical loads for UK mapping and recommended values for site-specific casework or Habitats Directive Article 17 reporting.

UK habitat	EUNIS class assigned	CL range (kg N ha^{-1} $year^{-1}$)	UK mapping value[a] (kg N ha^{-1} $year^{-1}$)	Recommended casework value (kg N ha^{-1} $year^{-1}$)
Saltmarsh	A2.54;A2.55;A2.53	20-30	25	20
Shifting coastal dunes	B1.3	10-20	n/a	10
Coastal dune grasslands	B1.4	8-15	9 (acid), 12 (non-acid)	8
Coastal dune heaths	B1.5	10-20	n/a	10
Moist to wet dune slacks	B1.8	10-20	n/a	10
Softwater lakes (oligotrophic)	C1.1	3-10	n/a	3
Dystrophic lakes/ponds/pools	C1.4	3-10	n/a	3
Raised & blanket bog	D1	5-10	8,9,10 (rainfall dependent)	5
Valley mires & poor fens	D2	10-15	n/a	10
Rich fens	D4.1	15-30	n/a	15
Montane rich fens	D4.2	15-25	n/a	15
Calcareous grassland	E1.26	15-25	15	15
Dry acid grassland	E1.7	10-15	10	10
Inland dune pioneer grassland	E1.94	8-15	n/a	8
Inland dune siliceous grassland	E1.95	8-15	n/a	8
Low & medium altitude hay meadows	E2.2	20-30	n/a	20
Mountain hay meadows	E2.3	10-20	n/a	10
Moist & wet oligotrophic grassland (Molinia)	E3.51	15-25	n/a	15
Wet acid grassland (Nardus & Juncus)	E3.52	10-20	15	10
Montane	E4.2	5-10	7	7
Alpine & subalpine acid & calcareous grassland	E4.3;E4.4	5-10	n/a	5
Arctic, alpine & subalpine scrub	F2	5-15	n/a	5
Wet dwarf shrub heath	F4.11	10-20	10	10
Dry dwarf shrub heath	F4.2	10-20	10	10
Broadleaved woodland[b]	G1	10-20	Included in G4	10
Beech woodland[b]	G1.6	10-20	15	15
Acidophilous oak woodland[b]	G1.8	10-15	10	10
Meso- & eutrophic oak woodland[b]	G1.A	15-20	n/a	15
Coniferous woodland[b]	G3	5-15	Included in G4	10
Scots Pine woodland[b]	G3.4	5-15	12	12
Broadleaf and/or conifer woodland[b][c]	G4	10-20, 5-15#	12	See G1 & G3

Table 1.1 footnotes:

(a) n/a where habitats not mapped nationally; for further information see Hall et al (2015a.)

(b) Unmanaged, non-productive woodland.

(c) Data are not available to separately map unmanaged broadleaved and unmanaged coniferous woodland nationally, with the exception of the woodlands in EUNIS classes G1.6, G1.8 and G3.4. Therefore G4 is used for all other unmanaged woodland mapped nationally.

\# Critical loads range 10-20 for broadleaf woodland and 5-15 for coniferous woodland

2. Work Package 2: Maps of pollutant concentrations and deposition

Summary

- *Concentration based estimated deposition (CBED) 5km data sets from 2004-06 onwards were updated in 2015 to correct for an over-estimation of nitric acid deposition.*
- *Long-term trends (1995 to 2014) in deposition budgets, based on CBED data to moorland and woodland, showed the largest reductions for non-marine sulphur (~64%) and a 15-20% reduction in total (oxidised plus reduced) nitrogen deposition.*
- *For calculating national-scale exceedances of critical levels of ammonia we recommend using FRAME 1km ammonia concentration data as this improves the spatial separation of ammonia source and sink areas.*

Pollutant concentration and deposition data used for this project were based on the Concentration Based Estimated Deposition (CBED) methodology. The CBED method generated 5km maps of wet and dry deposition of sulphur, oxidised and reduced nitrogen, and base cations, using measurements of air concentrations of gases and aerosols as well as concentrations in precipitation from the UK Eutrophying and Acidifying Pollutants (UKEAP) network. In addition, the FRAME model was used to generate 1km ammonia concentration data, and 5km deposition data for hindcast or future scenarios. This section summarises the concentration and deposition data used.

2.1 Deposition data

Under this contract CBED deposition data for the following years have been generated: 2005-07, 2007-09, 2008-10, 2009-11, 2010-12, 2011-13, 2012-14; data for 2006-08 were generated under the previous contract. This completed a sequence of rolling 3-year mean data sets from 1995 to 2014. There have been changes to the methods and data incorporated into the deposition data over this time period:

- Data for 1995-97, 1998-2000, 1999-2001 are based on the March 2004 version of CBED.
- Data for 2001-03 are based on the April 2005 version of CBED, which additionally included nitric acid; this is not included in the earlier data sets as the nitric acid network was not in operation prior to this date.
- All data from 2002-04 to 2012-14 are based on the May 2006 version of CBED, which additionally included aerosol deposition of ammonium, nitrate and sulphate, not available for the earlier data sets.

During 2014 it was discovered that nitric acid deposition had been overestimated. This was due to the current DELTA system (K2CO3-glycerol coated denuders) sampling other gas-phase oxidised nitrogen species in addition to nitric acid (Tang et al, 2015). The CBED data for the last 10 years (2004-2013) were re-calculated to correct for this over-estimation of nitric acid, and all subsequent data calculated incorporating the corrected method. The changes to the deposition budgets as a result of the over-estimation of nitric acid are shown in Figures 2.1 (deposition to moorland) and Figure 2.2 (deposition to woodland). The results showed a decrease in oxidised nitrogen (NOx) deposition of 13-16% for moorland and 27-29% for woodland, leading to a decrease in total nitrogen (NOx + NHx) deposition of 4-5% for moorland and 9-11% for woodland. There was no change to the reduced nitrogen (NHx) deposition. The largest decreases in nitrogen deposition were in the south-east of England (eg, Figure 2.3). The impact of the changes in deposition on critical loads exceedance are given in WP3.1.

Looking at the long-term trends in deposition budgets from 1995-97 to 2012-14 showed that non-marine sulphur (NMS) has decreased by 63%, NOx by 29-39% (deposition to moorland and woodland respectively) and NHx by 8% (Table 2.1). The differences in the NOx figures for the different habitats were because the wet NOx (wet + cloud droplet) changed by a different amount to the dry (dry + aerosols). Between 2004-06 and 2012-14 the wet dropped by about 13% for moorland and 15% for forest, while the dry dropped by 17% for moorland and 21% for forest. These drops reflect a reducing measured concentration but also different patterns of concentrations (a) between rainfall ion and gas, and (b) between years. Moorland and forest habitats get deposition in different fractions from the wet and dry components, and there were different spatial patterns of high and low concentrations in the maps. The differences were less obvious where there is little change (e.g. NHx) between years, or when the concentration maps show very similar structures for the years being compared (e.g. NMS).

Total nitrogen (NOx + NHx) and total acid (NOx + NHx + NMS) deposition maps based on the latest CBED data for 2012-14 are shown in Figure 2.4. These clearly show the enhanced deposition to woodland due to the higher dry deposition velocity for this habitat type. It should be noted that further changes to CBED deposition may be made in the future as a result of discussions to reduce the number of monitoring sites in the UKEAP network (Smith et al, 2014).

Table 2.1: Summary of reductions in deposition budgets from 1995-97 to 2012-14

Pollutant	% reduction in deposition budget to the UK 1995-97 to 2012-14	
	Deposition to moorland	Deposition to woodland
NMS	64%	63%
NOx	29%	39%
NHx	8%	8%
Total N	15%	20%

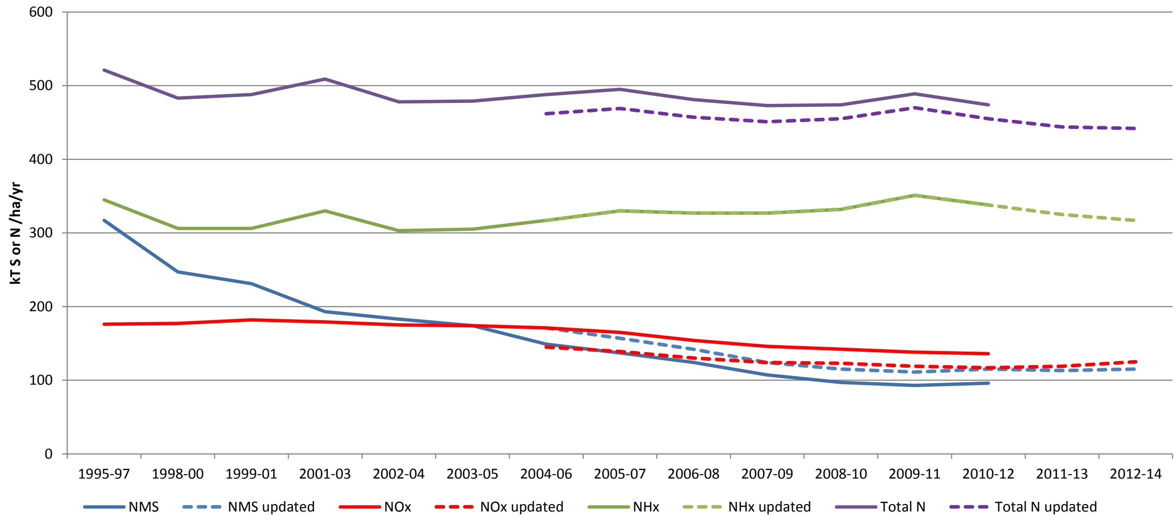


Figure 2.1: Budgets for CBED deposition to moorland; trends over time including updates to CBED 2004 to 2013.

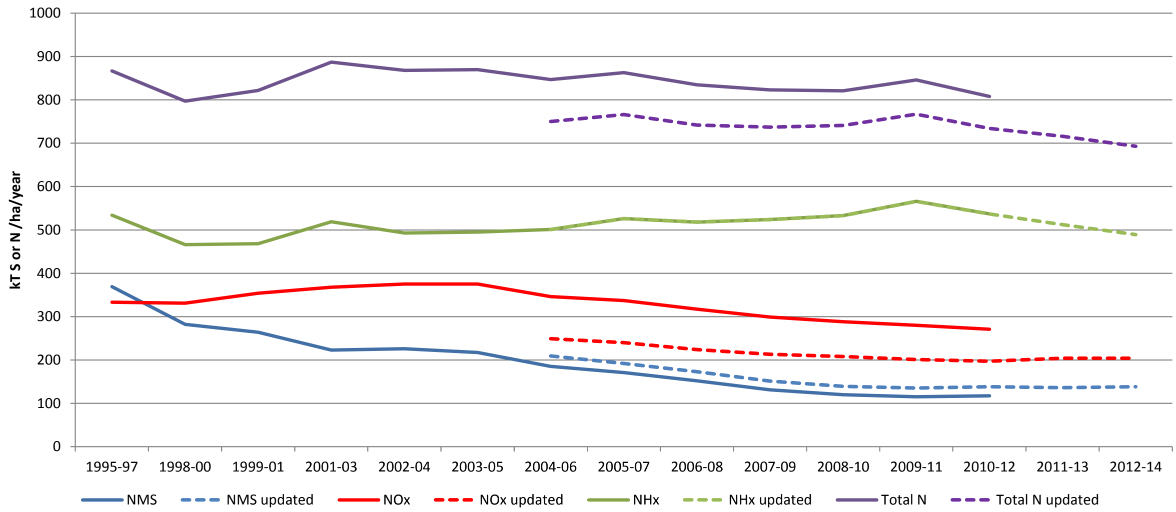


Figure 2.2: Budgets for CBED deposition to woodland; trends over time including updates to CBED 2004-2013.

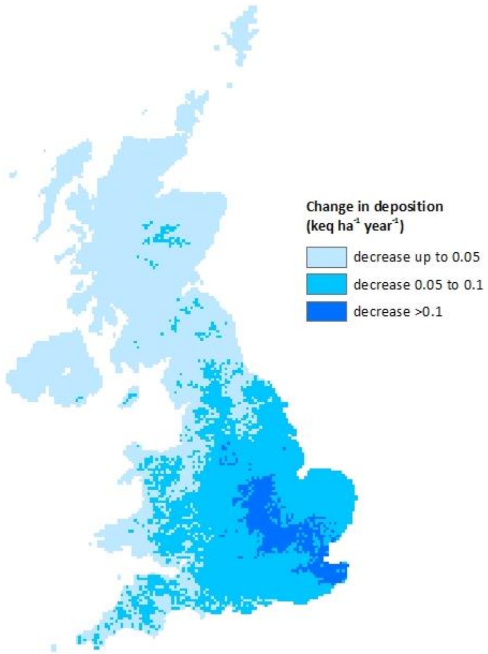


Figure 2.3: Map to show the spatial pattern of reductions in total nitrogen deposition to moorland for 2010-12, following the updates to the CBED methodology to correct for the over-estimate of nitric acid deposition. To convert from keq ha^{-1} $year^{-1}$ to kg N ha^{-1} $year^{-1}$ multiply by 14 (i.e. 0.1 keq ha^{-1} $year^{-1}$ = 1.4 kg N ha^{-1} $year^{-1}$.

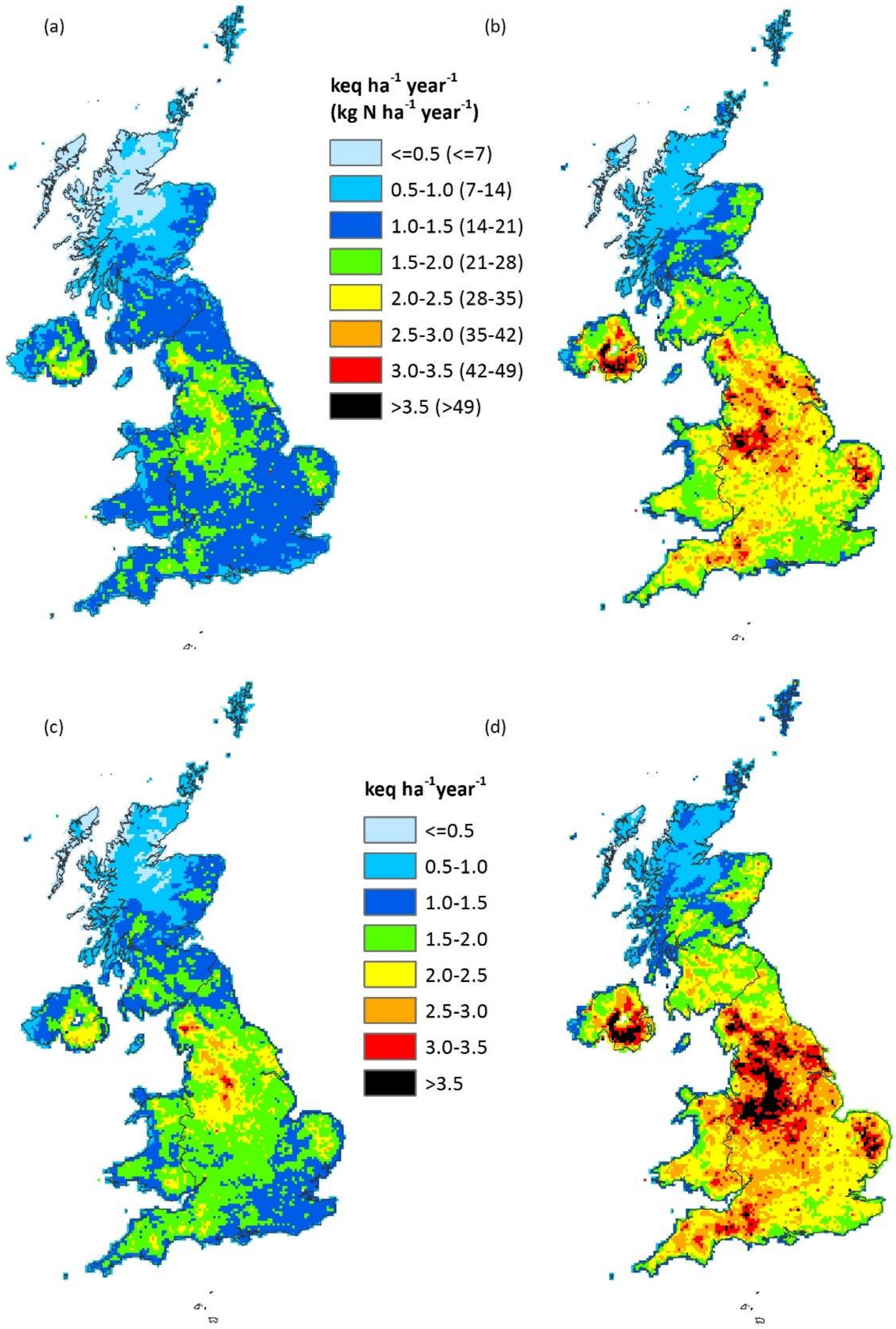


Figure 2.4: CBED 2012-14 total nitrogen deposition assuming (a) moorland everywhere, and (b) woodland everywhere, and total acid deposition assuming (c) moorland everywhere, and (d) woodland everywhere.

2.2 Concentration data

The site-based measurements from UKEAP were interpolated to generate 5km maps of concentrations of sulphur dioxide (SO_2), nitrogen oxides (NO_x), and ammonia (NH_3); data for 3-year rolling means for 2005-07, 2006-08, 2007-09, 2008-10, 2009-11, 2010-12 have been created during 2013 for this project. The CBED SO_2 and NO_x 5km concentration maps and exceedance results presented in this report have not been updated following the updates and corrections for nitric acid. The SO_2 and NO_x concentrations for all these years (Table 2.2) were below the critical levels for these pollutants (Table 3.10). However, these concentrations were based on the rural network only; including urban enhancement data from AEAT increased the concentrations around urban areas and around the major road networks (for NO_x)(Figure 2.5).

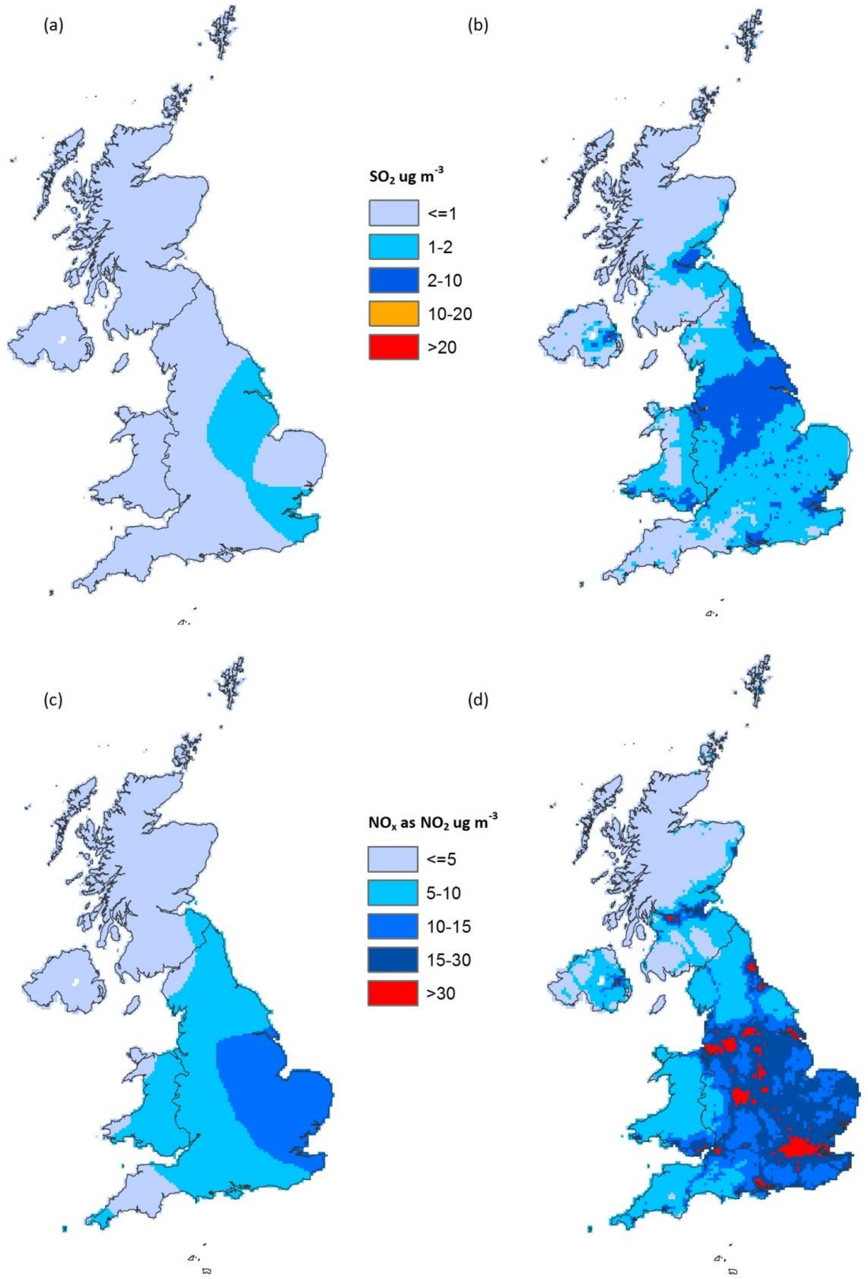


Figure 2.5: CBED 5km concentration data for 2010-12 (not updated): (a) SO_2 rural network only; (b) SO_2 rural plus urban enhancement; (c) NO_x rural network only; (d) NO_x rural plus urban enhancement.

Table 2.2: Summary of minimum, maximum and mean concentrations of CBED 5km 3-year mean data based on rural network data only.

Pollutant	Value	Concentrations in μg m^{-3} for 3-year mean datasets					
		2005-07	2006-08	2007-09	2008-10	2009-11	2010-12
SO_2	Min	0.18	0.16	0.16	0.13	0.12	0.11
	Max	1.99	2.03	1.89	1.59	1.39	1.25
	Mean	0.91	0.83	0.74	0.64	0.59	0.57
NO_x	Min	1.57	1.49	1.64	1.27	1.33	1.70
	Max	12.91	13.52	13.06	13.29	13.60	13.34
	Mean	6.38	6.22	6.27	6.11	6.09	5.98
NH_3	Min	0.01	0.01	0.01	0.01	0.01	0.01
	Max	9.63	11.90	12.67	13.00	14.07	12.97
	Mean	1.31	1.29	1.38	1.41	1.53	1.41

An alternative would be to use AEA-Ricardo Pollution Climate Mapping (PCM) 1km SO_2 and NO_x concentration data. These data highlight the higher concentrations in urban areas and for NO_x the higher concentrations along major transport routes (motorways and major roads)(Figure 2.6). Using these 1km data resulted in slightly larger areas exceeding the critical levels than the CBED 5km data, though the total areas exceeding the critical levels for SO_2 and NO_x across the UK were still small (Section 3.3).

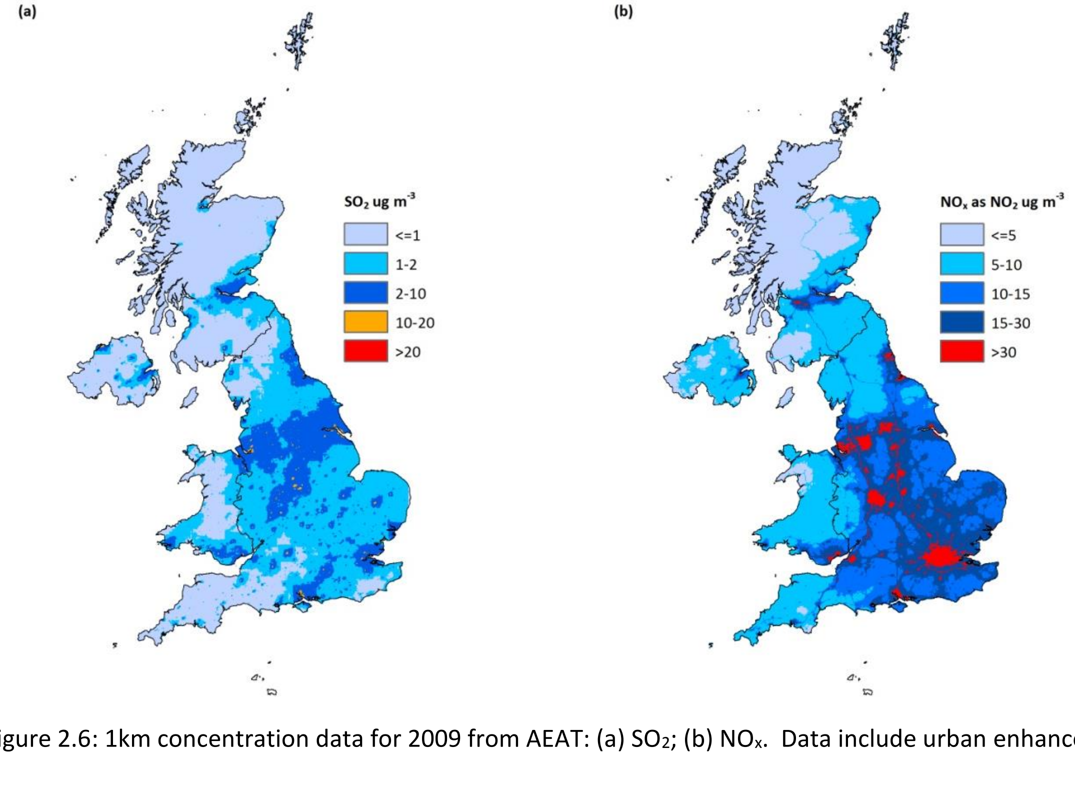


Figure 2.6: 1km concentration data for 2009 from AEAT: (a) SO_2; (b) NO_x. Data include urban enhancement.

For ammonia concentrations we recommend using 1km resolution data from the FRAME model; this resolution data has been found to be better at spatially separating the source (agricultural) areas from the sink areas (natural ecosystems) (Hallsworth et al, 2010). Defra funding has enabled us to develop an operational system based on FRAME that improves the spatial distribution of concentrations. The modelled NH_3 concentrations were calibrated relative to annually averaged measurements from the National Ammonia Monitoring Network using the median bias to adjust the

FRAME modelled concentrations. Data from all stations in the monitoring network were used for the calibration, with the exception of one station very close to a point source emitter that was not representative of the surrounding area. The FRAME NH_3 concentrations were updated annually, once the emissions data were available; the 2012 emissions data were available from the National Atmospheric Emissions Inventory web site in October 2014 and the NH_3 concentration data ready by December 2014. CBED also made use of the FRAME modelled NH_3 concentrations to calculate NH_x dry deposition at 5km resolution. As updates to the CBED 3-year mean deposition data were required earlier than this, CBED used the NH_3 modelled concentrations from the previous year, for example, in the CBED 2011-2013 deposition data set, deposition for the year 2013 used the 2012 NH_3 concentrations calibrated relative to measurements for the year 2013.

Using 5km resolution NH_3 concentration data can over-estimate the area of natural ecosystems exceeding the critical levels (Figure 2.7 and see Section 3.3.3), due to the spatial mixing of sources and sinks within an individual grid square. Using the 1km ammonia concentration data is consistent with the approach taken in the Defra funded "Ammonia for future patterns" project (AC0109) and with the approach used in the Defra funded Pollution Climate Mapping model to calculate a range of gas and particulate concentrations distributed at a national scale.

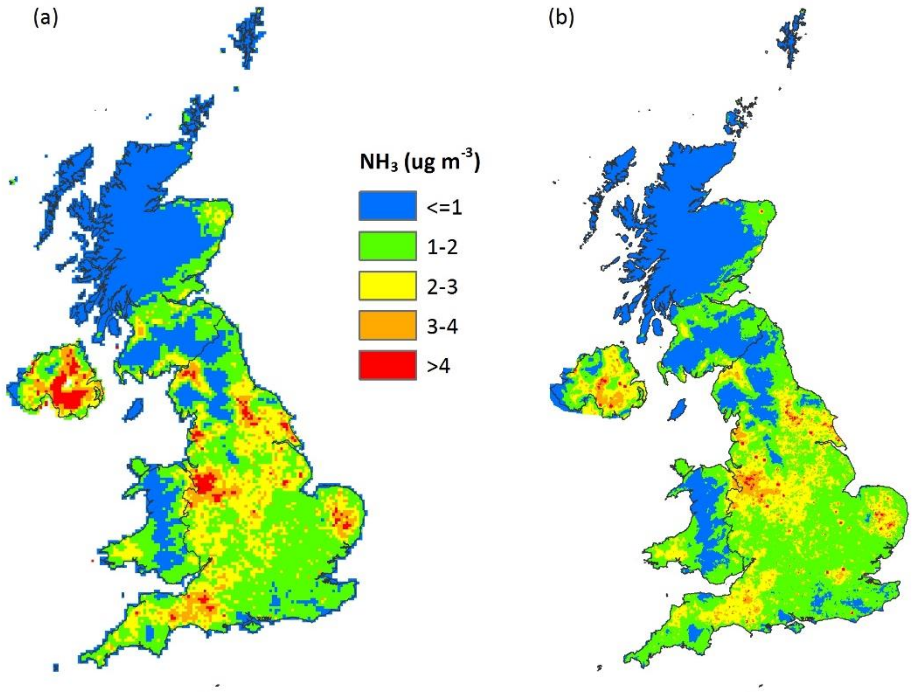


Figure 2.7: Mean NH_3 concentrations for 2010-12 at (a) 5km resolution from CBED, and (b) 1km resolution from FRAME.

FRAME ammonia concentrations are currently available for the years, 2009-11, 2010-12 and 2011-13 (Figure 2.8); exceedances of the ammonia critical levels based on these concentration data are summarised in Section 3.3.5

The issue of grid resolution is different for deposition, as rainfall makes a high contribution to sulphur and nitrogen deposition across the country. Considerable spatial heterogeneity in rainfall within a 5km area is evident in mountainous areas (ie, Dore et al. 2006, 2012). However due the low number of rain gauges in the UK Meteorological Office network located in upland regions, there is high uncertainty associated with calculating annual precipitation in these areas. Furthermore measurements of rainfall in the uplands are particularly subject to error due to under-capture of precipitation in wind exposed locations and not registering precipitation which falls as snow in sub-zero temperatures. For this reason, deposition data calculated with CBED will continue to be calculated at a 5km resolution.

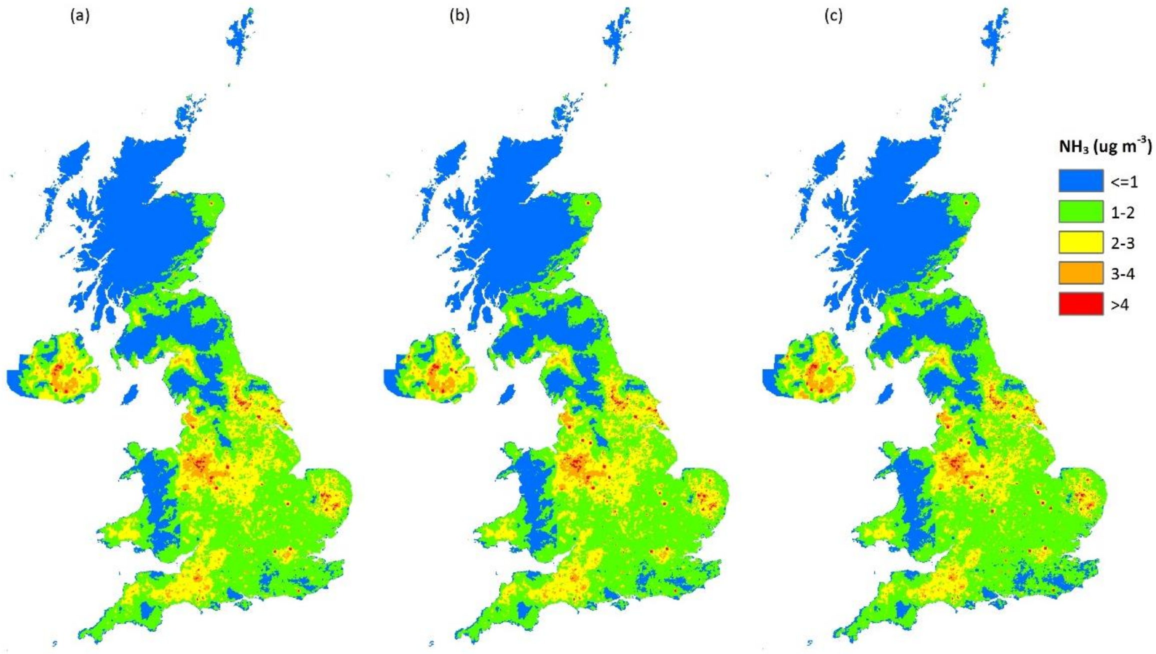


Figure 2.8. FRAME mean ammonia concentrations for (a) 2009-2011; (b) 2010-2012; (c) 2011-2013.

3. Work Package 3: Exceedance data and indicators

Summary

- *Summary critical load exceedance statistics have been updated using CBED deposition data to provide long-term trends in exceedances from 1995 to 2014; results were reported to Defra for use as an air pollution indicator for biodiversity (http://jncc.defra.gov.uk/page-1824).*
- *The latest national habitat critical load exceedance results based on CBED deposition data for 2012-14 showed 44.1% of the UK area of acid-sensitive habitats have exceedance of acidity critical loads, and an average accumulated exceedance (AAE) of 0.28 keq ha^{-1} $year^{-1}$. For nutrient nitrogen, critical loads were exceeded across 62.2% of the UK area of nitrogen-sensitive habitats, with an AAE of 6.0 kg N ha^{-1} $year^{-1}$.*
- *The long-term trends (1995 to 2014) in critical load exceedance showed a 28.5% reduction in the area of UK acid-sensitive habitats with exceedance of acidity critical loads and a reduction of 64% in the AAE. For nutrient nitrogen the reductions were smaller, reflecting the smaller reductions in nitrogen deposition compared to acid deposition, with a 12.8% reduction in the UK area of nitrogen-sensitive habitats with critical load exceedance and a 36.7% reduction in AAE.*
- *The latest UK SRCL exceedance results based on CBED deposition for 2012-14 showed that 76% of SACs, 70.3% of SPAs and 61.4% of SSSIs had exceedance of acidity critical loads for at least one feature in a site. Nutrient nitrogen critical loads were exceeded for at least one feature in 90.1% of SACs, 73.3% of SPAs and 88.1% of SSSIs.*
- *The long-term trends (1995 to 2014) in exceedances of SRCL for UK SACs, SPAs and SSSIs showed a reduction in the percentage of sites with at least one feature exceeded, of 15-23% for acidity and 5-8% for nutrient nitrogen.*
- *Ammonia concentrations (FRAME 1km data) for 2011-13 exceeded the critical level of 1µg m^{-3} across 62.7% of the UK land area; only 3.8% of the UK land area received concentrations above the critical level of 3 µg m^{-3}. The percentage of UK SACs with ammonia concentrations exceeding the critical levels anywhere across a site, was 60.9% for the critical level of 1µg m^{-3} and 7.5% for the critical level of 3 µg m^{-3}.*

"Exceedance" refers to the amount of excess deposition above the critical load, or the concentration above the critical level. This section describes the exceedance metrics calculated and presents summary results of critical loads and critical levels exceedances.

3.1 Exceedance of critical loads for UK habitats sensitive to acidification and/or eutrophication

The exceedance calculations were carried out using the spatial data stored in ArcGIS and a suite of Python scripts; these have been developed and further updated to calculate the following exceedance metrics for acidity and for nutrient nitrogen by habitat and country:

- Habitat area exceeded and percentage habitat area exceeded; this is a useful metric but it can be insensitive to changes between years or scenarios, as the area exceeded can remain the same even if there is a change in the magnitude of exceedance.
- Accumulated Exceedance (AE in keq $year^{-1}$) = exceedance (keq ha^{-1} $year^{-1}$) * exceeded area (ha)
  AE can be summarised across large areas and can be useful for comparing results between years or scenarios, but the numbers are very large and not intuitive to understand; the same value can result for large exceedances and small areas or small exceedances and large areas.
- Average Accumulated Exceedance (AAE in keq ha^{-1} $year^{-1}$) = AE / total habitat area (ha)

This metric averages the exceedance across the entire habitat area and can be used to give an indication of the change in the magnitude of exceedance even if the area exceeded remains the same; this metric is also useful for providing summary maps for all habitats combined.

In addition new scripts have been incorporated that automatically generate critical load exceedance maps in pdf and jpeg formats:

- Exceedance of 5th-percentile critical loads for all terrestrial habitats combined*
- Average Accumulated Exceedance for all terrestrial habitats combined*

*The maps represent the exceedances for all terrestrial habitats combined into a single map, based on the exceedance results for all habitats mapped nationally as sensitive to acidification and/or eutrophication. Areas of land occupied by other habitats types are left blank (white) on the maps. In addition, it should be noted that these exceedance maps exclude the results for freshwaters because the data for waters are based on catchment areas rather than 1km grid squares, and as such may overlap with other habitat data. Separate maps of the acidity critical load exceedances for the freshwater sites included in national database are generated and stored within ArcGIS.

At the European scale maps of AAE are now used more frequently than those based on percentile critical loads (eg, see Slootweg et al, 2014); one advantage is that they include the exceedances of all the critical loads for all habitats without creating a summary statistic (and deciding what that should be) of the critical loads data first. This report only includes UK exceedance maps based on AAE as they provide a better representation of the summary critical load exceedance statistics than the exceedance maps based on percentile critical loads; differences between maps of percentile critical load exceedances and AAE maps are described in the Methods Report (Hall et al, 2015a).

The trends in the percentage habitat area exceeded for acidity and for nutrient nitrogen have been used by Defra and JNCC as an air pollution indicator for biodiversity (http://jncc.defra.gov.uk/page-1824). The exceedance results for 2009-11 and 2010-12 were submitted to Defra in May 2013 and May 2014 respectively, for inclusion in the updates to the indicator.

Exceedances were calculated using rolling 3-year mean CBED deposition data sets. Results have been generated under this contract for the years 2005-07, 2007-09, 2008-10, 2009-11, and 2010-12 using the May 2006 version of CBED (Section 2.1). Data for 2006-08 were already available under the previous contract. Following the recent updates to CBED to correct for the over-estimate of nitric acid deposition (Section 2.1), exceedances have been re-calculated for the years 2004-06, 2005-07, 2006-08, 2007-09, 2008-10, 2009-11 and 2010-12 and the two sets of exceedance results compared. In addition exceedances have been calculated using the new CBED data for 2011-13 and 2012-14. At the UK level, the changes to CBED deposition had little effect on the percentage area of habitats exceeding critical loads (Figure 3.1); differences in the magnitude of exceedance, expressed as AAE, were between 0.02-0.05 keq ha^{-1} year^{-1} for acidity, and 0.07 to 0.1 keq ha^{-1} year^{-1} (equivalent to 0.98 to 1.4 kg N ha^{-1} year^{-1}) for nutrient nitrogen (Figure 3.2).

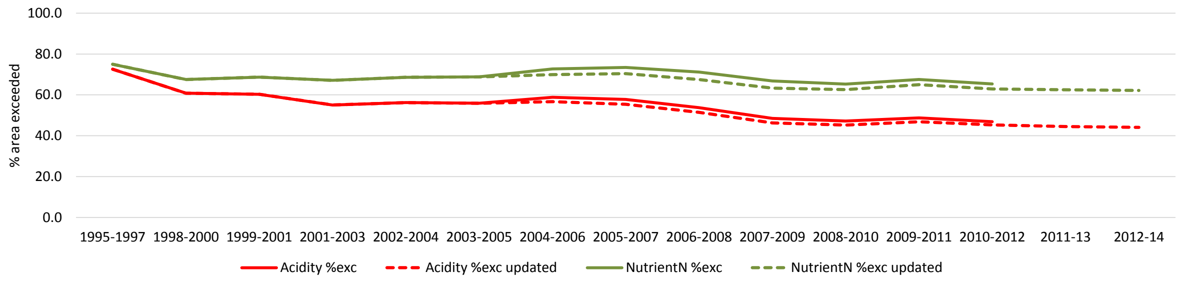


Figure 3.1: Percentage area of habitats exceeding critical loads of acidity and of nutrient nitrogen, comparing the original results (solid line) with those based on the updated CBED deposition (dashed line).

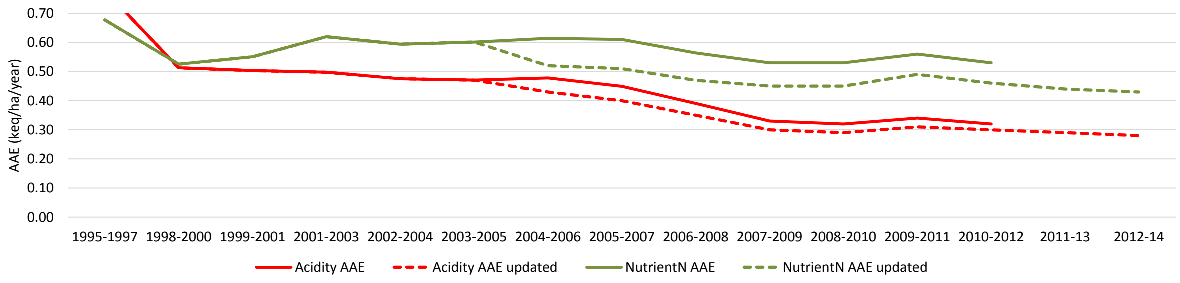


Figure 3.2: AAE of acidity and nutrient nitrogen critical loads, comparing the original results (solid line) with those based on the updated CBED deposition (dashed line).

The updated trends in the percentage areas of UK acid- and nitrogen-sensitive habitats with exceedance of critical loads were provided to Defra and JNCC in August 2015 for updating the air pollution indicator for biodiversity (http://jncc.defra.gov.uk/page-1824), together with an explanatory note giving the reason for the changes (ie, changes made to CBED deposition). The trends in exceedances were also made available on the project website (www.cldm.ceh.ac.uk) and in the annually updated "Trends Report" (Hall et al, 2015b, 2016) also available from the project website. If in the future changes are made to the number of sites included in the UKEAP monitoring network (Section 2.1), the trends in critical load exceedances may require a further update, depending upon the impact such changes have on the spatial patterns and values on the CBED deposition maps.

Tables 3.1 and 3.2 present the updated trends in exceedances by country and Figures 3.3 and 3.4 present the UK trends in exceedances by habitat (Figures 3.3 and 3.4). They show for acidity that the percentage area of acid-sensitive habitats in the UK with exceedance of critical loads decreased by 28.5% (72.6% to 44.1%) from 1995-97 to 2012-14, and AAE reduced by 64% (0.78 to 0.28 keq ha^{-1} $year^{-1}$ over the same time span. Changes in exceedances were smaller for nitrogen, reflecting the smaller changes in nitrogen deposition compared to sulphur, with a 12.8% reduction (75% to 62.2%) in the area of nitrogen-sensitive habitats in the UK exceeded and a 36.7% reduction in the AAE, equivalent to an average decrease in exceedance across the UK of 3.5 kg N ha^{-1} $year^{-1}$. The results for 2011-13 and 2012-14 were similar for all countries except NI, which showed a 6% reduction in the area exceeded; this was consistent with reductions of 3-5 kT N/year in the ammonia deposition budgets to moorland and woodland respectively. Although these reductions in deposition budgets were similar to those elsewhere in the UK, as NI is relatively small, these changes were large enough to reduce the areas exceeding critical loads.

The nitrogen results for the UK (Figure 3.4) showed that for some habitats, particularly woodlands, there was little or no change in the total habitat area exceeding nutrient nitrogen critical loads, but there were reductions in the AAE. The average AAE for nitrogen for all habitats across the UK, based on 2012-14 deposition was 6.2 kg N ha^{-1} $year^{-1}$; but this varied spatially as can be seen in Table 3.2 and Figure 3.5b. The magnitude of exceedance was generally lower for acidity (Figure 3.5a). The spatial patterns of exceedance (Figure 3.5) were consistent with the spatial patterns of deposition (Figure 2.4).

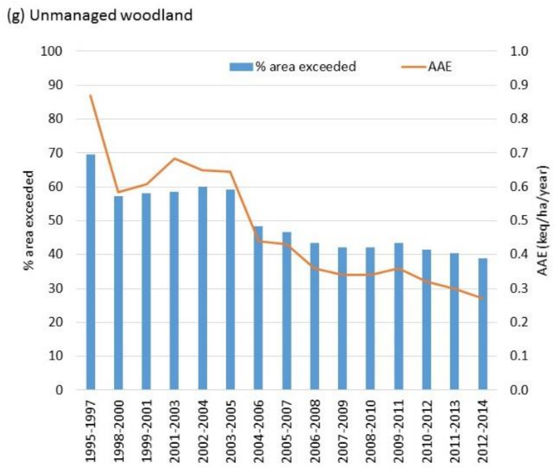
Table 3.1: Acidity: Trends by country in (a) the percentage area of acid-sensitive habitats where critical loads are exceeded; (b) Average Accumulated Exceedance

Deposition dataset	Percentage area of sensitive habitats with exceedance of acidity critical loads					Acidity Average Accumulated Exceedance (AAE: keq ha^{-1} $year^{-1}$)				
	England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
1995-1997	75.8	90.0	68.2	76.8	72.6	1.33	1.36	0.47	0.80	0.78
1998-2000	71.6	83.1	52.6	67.2	60.8	1.00	0.84	0.28	0.46	0.51
1999-2001	71.9	83.0	51.6	66.8	60.3	0.98	0.82	0.27	0.46	0.50
2001-2003	72.3	82.4	43.0	67.4	55.0	1.04	0.82	0.23	0.51	0.50
2002-2004	72.3	82.3	44.8	69.2	56.2	0.94	0.79	0.24	0.46	0.48
2003-2005	71.8	83.2	44.5	67.1	55.9	0.93	0.84	0.24	0.42	0.47
2004-2006	66.8	81.2	48.0	68.1	56.7	0.77	0.74	0.24	0.42	0.43
2005-2007	66.1	81.0	46.1	68.5	55.4	0.74	0.73	0.21	0.45	0.40
2006-2008	64.3	79.2	40.7	68.6	51.4	0.68	0.61	0.17	0.44	0.35
2007-2009	63.6	77.4	32.9	69.4	46.3	0.62	0.54	0.12	0.45	0.3
2008-2010	63.2	74.9	31.5	69.6	45.2	0.59	0.49	0.12	0.47	0.29
2009-2011	63.8	74.5	33.9	71.0	46.8	0.62	0.48	0.15	0.53	0.31
2010-2012	62.8	74.2	32.2	67.8	45.3	0.6	0.47	0.14	0.46	0.3
2011-2013	62.1	74.4	31	69.4	44.5	0.59	0.47	0.13	0.46	0.29
2012-2014	61.6	75.3	30.9	63.4	44.1	0.56	0.51	0.13	0.35	0.28

Table 3.2: Nutrient nitrogen: Trends by country in (a) the percentage area of nitrogen-sensitive habitats where critical loads are exceeded; (b) Average Accumulated Exceedance

Deposition dataset	Percentage area of sensitive habitats with exceedance of nutrient nitrogen critical loads					Nutrient nitrogen Average Accumulated Exceedance (AAE: kg N ha^{-1} $year^{-1}$)				
	England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
1995-1997	98.3	98.0	59.4	92.6	75.0	19.0	15.8	4.1	10.6	9.5
1998-2000	97.6	92.5	48.9	80.0	67.5	16.8	10.3	2.7	6.5	7.4
1999-2001	97.7	91.1	50.9	82.5	68.7	17.4	10.6	2.9	6.8	7.7
2001-2003	97.8	93.5	47.7	85.4	67.1	19.7	12.2	3.1	8.9	8.7
2002-2004	97.6	93.3	50.2	86.3	68.6	18.0	12.2	3.3	8.7	8.3
2003-2005	97.5	94.1	50.6	83.8	68.8	18.2	13.2	3.3	8.3	8.4
2004-2006	96.7	93.2	52.9	84.8	69.9	14.9	11.4	3.1	7.9	7.2
2005-2007	96.5	93.6	53.6	86.4	70.4	14.9	11.4	2.9	8.8	7.2
2006-2008	96.1	92.9	49.0	86.8	67.5	14.1	9.9	2.5	8.8	6.6
2007-2009	96.4	91.7	41.8	88.7	63.3	13.8	9.5	2.1	9.4	6.3
2008-2010	96.5	89.7	40.7	89.7	62.6	13.9	9.2	2.2	9.8	6.3
2009-2011	97.0	89.8	44.5	91.4	65.0	14.6	9.2	2.6	10.9	6.8
2010-2012	96.5	89.6	41.4	88.5	62.9	13.8	8.8	2.4	9.6	6.4
2011-2013	96.0	90.3	40.7	89.9	62.5	13.3	8.9	2.3	9.5	6.2
2012-2014	96.1	90.9	40.7	83.0	62.2	12.7	9.1	2.3	7.6	6.0

Figure 3.3 (a-f): Acidity: UK results of the percentage area of acid-sensitive habitats* with exceedance of critical loads and the Average Accumulated Exceedance (keq ha^{-1} $year^{-1}$). For information on the CBED deposition data used refer to Section 2.1. * "Freshwaters" results were based on data for 1752 freshwater catchments in the UK. For further information please refer to Hall et al, 2015a.



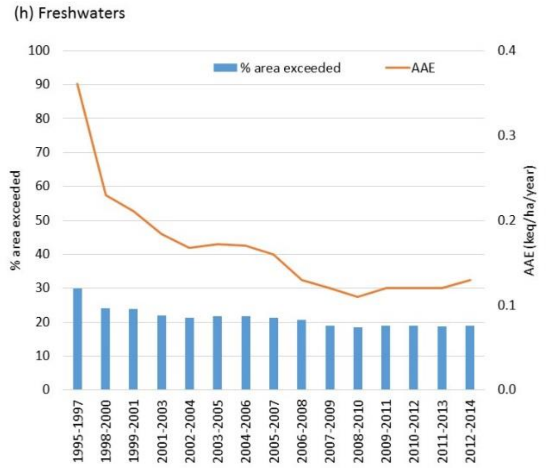


Figure 3.3 (g-h): Acidity: UK results of the percentage area of acid-sensitive habitats* with exceedance of critical loads and the Average Accumulated Exceedance (keq ha^{-1} $year^{-1}$). For information on the CBED deposition data used refer to Section 2.1. * "Freshwaters" results were based on data for 1752 freshwater catchments in the UK. For further information please refer to Hall et al, 2015a.

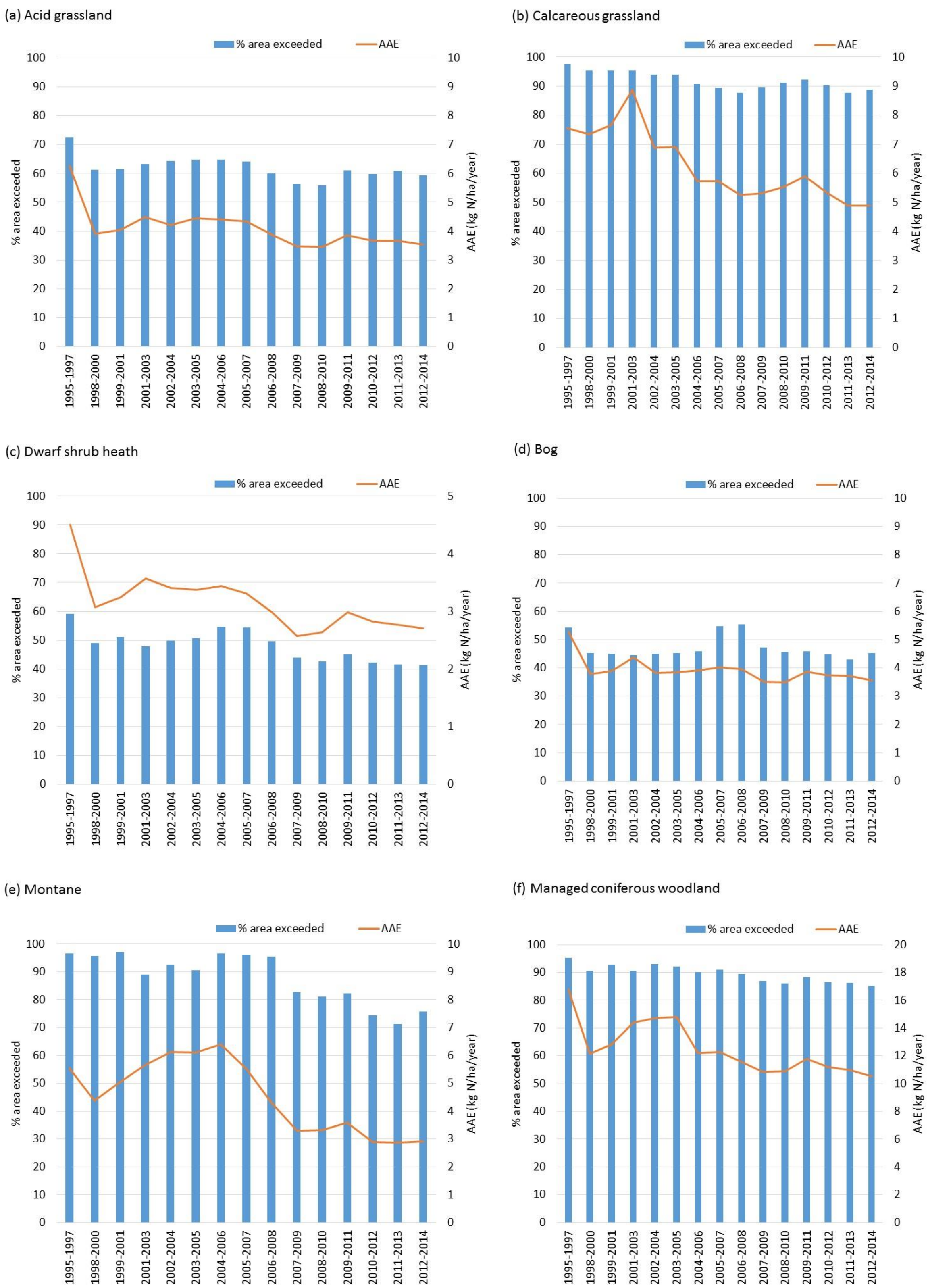


Figure 3.4 (a-f). Nutrient nitrogen: UK results of the percentage area of nitrogen-sensitive habitats* with exceedance of critical loads and the Average Accumulated Exceedance (kg N ha^{-1} $year^{-1}$). For information on the CBED deposition data used refer to Section 2.1. * Results for Saltmarsh habitat not shown; area of habitat exceeded by CBED deposition in 1995-97 was 2% (0.7% with data for 2012-14), and the AAE was <0.1 kg N ha^{-1} $year^{-1}$ in most years.

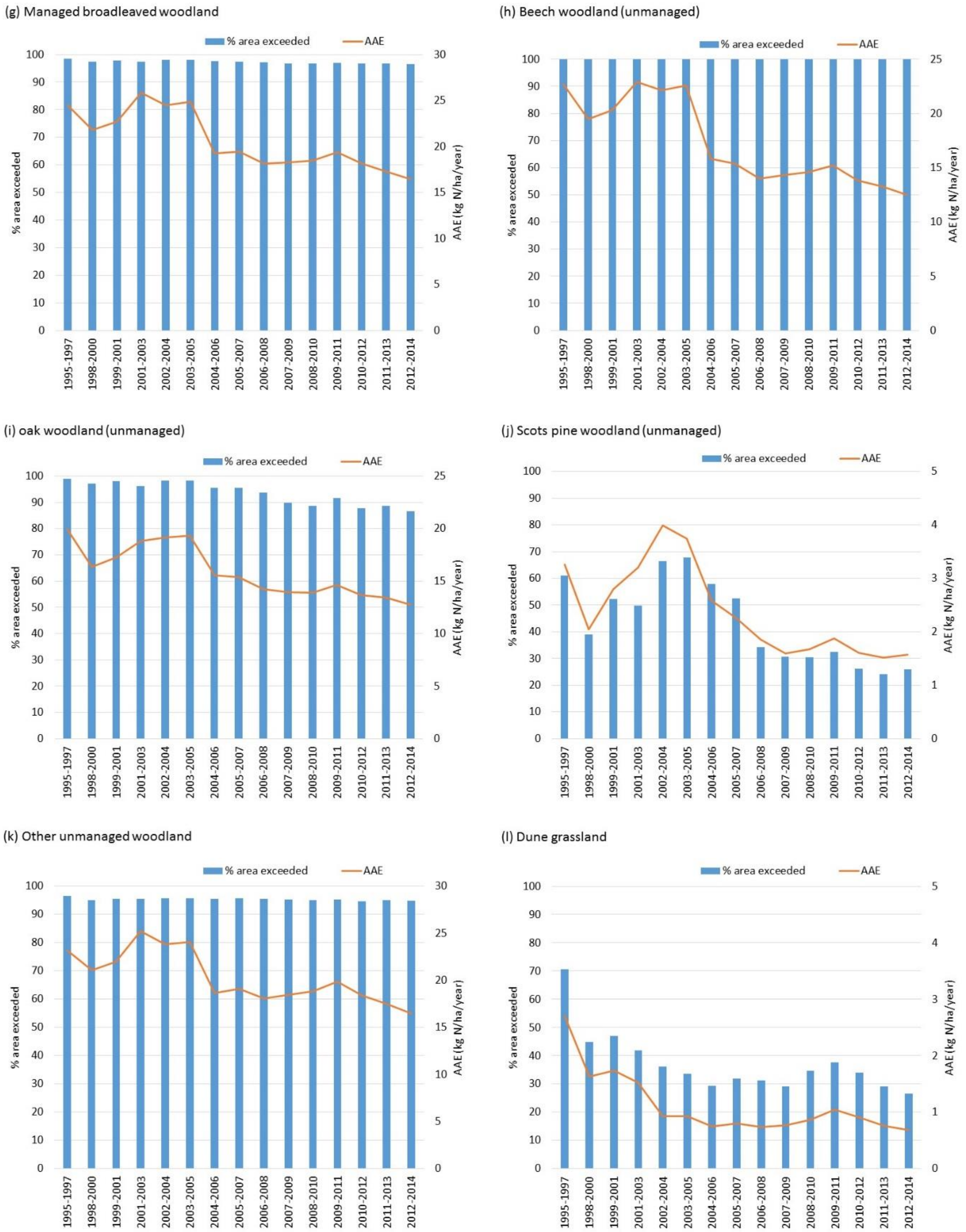


Figure 3.4 (g-l). Nutrient nitrogen: UK results of the percentage area of nitrogen-sensitive habitats* with exceedance of critical loads and the Average Accumulated Exceedance (kg N ha^{-1} $year^{-1}$). For information on the CBED deposition data used refer to Section 2.1. * Results for Saltmarsh habitat not shown; area of habitat exceeded by CBED deposition in 1995-97 is 2% (0.7% with data for 2012-14), and the AAE is <0.1 kg N ha^{-1} $year^{-1}$ in most years.

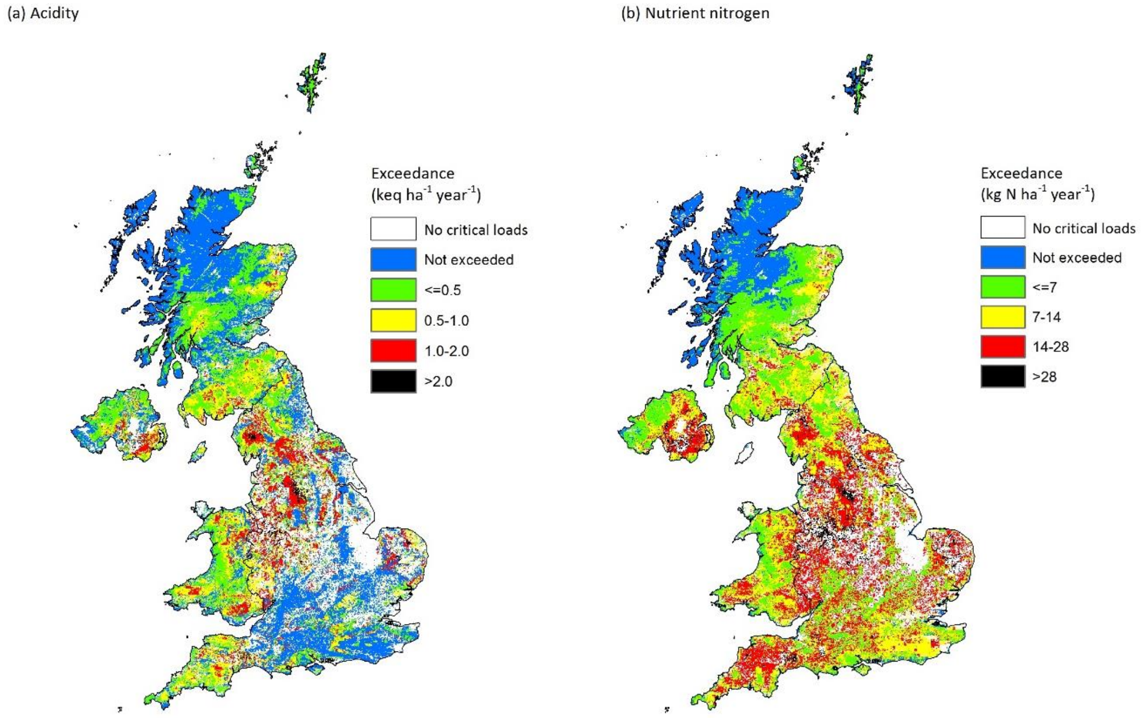


Figure 3.5: Average Accumulated Exceedance (AAE) of critical loads by CBED deposition for 2012-14. (a) Acidity; (b) Nutrient nitrogen. Note that although the legends are presented in different units for acidity and for nutrient nitrogen, the class intervals are equivalent (eg, 7 kg N ha^{-1} $year^{-1}$ = 0.5 keq ha^{-1} $year^{-1}$). The maps represent the exceedances for all terrestrial habitat types mapped nationally as sensitive to acidification and/or eutrophication (WP1); areas of the UK containing other habitats to which critical loads have not been applied are shown in white.

3.2 Exceedance of Site Relevant Critical Loads (SRCL)

A separate suite of Python scripts has been developed to automate the SRCL exceedance calculations, generation of summary exceedance statistics and exceedance maps. The scripts generated the following summary information separately for SACs, SPAs and SSSIs by country:

- Number and percentage of sites where the critical load for any designated feature habitat was exceeded
- Maximum area of sites with features exceeding critical loads
- Maximum Accumulated Exceedance (AE)
- Maximum Average Accumulated Exceedance (AAE)

The scripts also output the following information for each site:

- Exceedance for each feature habitat
- Maximum exceedance of any feature
- Maximum exceeded area for any feature*
- Maximum AE of any feature
- Maximum AAE of any feature

*As the spatial location of the designated features is not available in digital format, the feature area is assumed to be the same as the site area; the exceeded area is calculated as the sum of the 1km squares or parts thereof within the site where the critical loads are exceeded.

Five different exceedance maps were created:

(i) Total number of features per site
(ii) Sites where any feature is exceeded
(iii) Number of exceeded features per site
(iv) Percentage of features exceeded per site
(v) Maximum AAE (of any feature) per site

A brief overview of the latest exceedance results for UK SACs, SPAs and SSSIs, based on CBED deposition for 2012-14 is given in Tables 3.5 and 3.6 below. Detailed results will be included in a future update of the Trends Report (Hall et al, 2016) and made available via the project website (www.cldm.ceh.ac.uk).

Table 3.5: Summary acidity critical load exceedance results for UK designated sites based on CBED deposition for 2012-14.

Site type	No. of sites in the UK	No. sites with critical loads#	Percentage of sites with exceedance of critical loads for at least one feature##	Maximum AAE of all sites/features (keq ha^{-1} yr^{-1})
SACs	616	487	76.0	0.66
SPAs	257	175	70.3	0.46
SSSIs	6876	4683	61.4	0.48

#Number of sites with critical loads assigned to at least one habitat feature.
##Calculated as a percentage of the number of sites with critical loads.

Table 3.6: Summary nutrient nitrogen exceedance results for UK designated sites based on CBED deposition for 2012-14

Site type	No. of sites in the UK	No. sites with critical loads#	Percentage of sites with exceedance of critical loads for at least one feature##	Maximum AAE of all sites/features (kg N ha^{-1} yr^{-1})
SACs	616	536	90.1	9.1
SPAs	257	225	73.3	8.7
SSSIs	6876	4521	88.1	9.7

#Number of sites with critical loads assigned to at least one habitat feature.
##Calculated as a percentage of the number of sites with critical loads.

Following the updates to the “recommended” nutrient nitrogen critical loads (Section 1.2), trends in critical load exceedances for SRCL were re-calculated using the CBED deposition data for the years 1995-97 to 2010-14 (Section 2.1). Table 3.7 summarises the reductions in the percentage of sites with exceedance of at least one feature habitat, and the associated percentage reductions in the maximum AAE.

Further results in the trends in critical load exceedances for UK SACs are given below; results for SPAs and SSSIs will be included in a future update of the Trends Report (Hall et al, 2015b) and made available via the project website (www.cldm.ceh.ac.uk).

Table 3.7: Summary of the percentage reductions in the percentage of sites with the critical loads for at least one feature exceeded, and the reductions in maximum AAE (of all sites/features) from 1995 to 2014.

Site type	Acidity results: % reduction in:		Nutrient nitrogen results: % reduction in:	
	No. sites exceeded	AAE	No. sites exceeded	AAE
SACs	15%	56%	5%	35%
SPAs	23%	59%	8%	35%
SSSIs	16%	59%	7%	35%

For SACs, between 1995-97 and 2012-14 the percentage of sites in the UK with exceedance of acidity critical loads for one or more features decreased by 15% and the maximum AAE decreased by 56% (0.85 keq ha^{-1} year^{-1})(Table 3.8). For acidity, the largest decrease in the percentage of sites with exceedance was in Scotland (down 28%); this was accompanied by a 71% decrease in maximum AAE, however, the AAE values for Scotland were much lower than in other parts of the UK in all years (due to lower deposition in Scotland compared to other areas of the UK). For nutrient nitrogen, temporal changes were smaller due to the smaller reductions in nitrogen deposition, compared to acidity; there were 4.9% less sites with exceedance of critical loads (for one or more features) in 2012-14 compared to 1995-97, with a reduction in the maximum AAE of 5 kg N ha^{-1} year^{-1} (Table 3.9). Decreases in the percentage of sites with any feature exceeded were similar for England, Wales and Scotland (4.1-6.5%). Results for Northern Ireland remained virtually unchanged in all years with the majority of sites having exceedance for at least one habitat feature in all years. Maps showing the latest exceedance results for SACs for 2012-14 are shown in Figure 3.8.

Table 3.8: Acidity results: trends in the percentage of SACs (with acidity critical loads) with exceedance of one or more feature habitats, and maximum AAE.

Deposition dataset	Percentage of sites with exceedance of acidity critical loads for at least one feature					Maximum acidity Average Accumulated Exceedance (AAE: keq ha^{-1} year^{-1})				
	England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
1995-1997	85.0	97.2	92.3	97.9	91.0	2.36	1.87	0.66	1.32	1.51
1998-2000	82.2	97.2	83.5	95.7	86.4	1.80	1.29	0.42	0.76	1.10
1999-2001	81.7	97.2	83.5	95.7	86.2	1.83	1.31	0.44	0.78	1.12
2001-2003	81.1	94.4	75.3	95.7	82.5	1.89	1.31	0.41	0.87	1.13
2002-2004	82.8	95.8	78.0	95.7	84.4	1.77	1.27	0.43	0.77	1.09
2003-2005	82.8	95.8	76.4	95.7	83.8	1.75	1.33	0.42	0.71	1.08
2004-2006	79.4	95.8	79.7	95.7	83.8	1.50	1.08	0.42	0.70	0.95
2005-2007	79.4	95.8	79.7	95.7	83.8	1.45	1.05	0.38	0.73	0.91
2006-2008	77.2	95.8	75.8	95.7	81.5	1.35	0.90	0.31	0.71	0.82
2007-2009	76.7	95.8	69.2	95.7	78.9	1.21	0.82	0.22	0.72	0.71
2008-2010	75.6	95.8	67.6	95.7	77.8	1.16	0.77	0.22	0.75	0.68
2009-2011	76.1	95.8	70.3	95.7	79.1	1.20	0.75	0.23	0.79	0.71
2010-2012	76.1	93.0	68.1	93.6	77.6	1.17	0.75	0.21	0.72	0.68
2011-2013	75.0	93.0	68.1	95.7	77.4	1.18	0.75	0.19	0.72	0.67
2012-2014	74.4	94.4	64.3	95.7	76.0	1.14	0.79	0.19	0.63	0.66

Table 3.9: Nutrient nitrogen results: trends in the percentage of SACs (with nutrient nitrogen critical loads) with exceedance of one or more feature habitats, and maximum AAE.

Deposition dataset	Percentage of sites with exceedance of nutrient nitrogen critical loads for at least one feature					Maximum nutrient nitrogen Average Accumulated Exceedance (AAE: kg N ha^{-1} $year^{-1}$)				
	England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
1995-1997	98.5	98.7	89.6	98.0	94.96	20.51	14.07	7.27	14.41	14.06
1998-2000	97.0	96.2	85.1	96.0	92.16	17.36	10.25	5.85	9.76	11.45
1999-2001	97.0	96.2	85.6	96.0	92.35	18.33	10.87	6.32	10.32	12.15
2001-2003	98.0	94.9	84.6	98.0	92.35	19.76	11.85	6.25	12.73	12.95
2002-2004	97.5	93.7	85.6	98.0	92.35	18.37	11.71	6.59	11.33	12.44
2003-2005	97.5	96.2	85.6	98.0	92.72	18.63	12.23	6.40	10.89	12.53
2004-2006	95.9	94.9	84.6	98.0	91.60	15.76	9.71	6.24	11.05	10.89
2005-2007	94.9	94.9	86.1	98.0	91.79	15.72	9.74	6.51	11.94	11.02
2006-2008	94.4	93.7	86.6	98.0	91.60	14.97	8.75	6.08	11.85	10.38
2007-2009	94.9	93.7	83.1	98.0	90.49	14.13	8.51	5.10	12.29	9.62
2008-2010	95.4	93.7	82.6	98.0	90.49	13.98	8.37	4.82	12.66	9.43
2009-2011	95.9	93.7	84.1	98.0	91.23	14.54	8.36	4.93	13.10	9.73
2010-2012	95.4	93.7	83.1	98.0	90.67	14.03	8.15	4.59	12.22	9.31
2011-2013	93.9	93.7	82.6	98.0	89.93	13.92	8.19	4.44	12.23	9.22
2012-2014	94.4	93.7	83.1	96.0	90.11	13.53	8.32	4.56	11.11	9.08

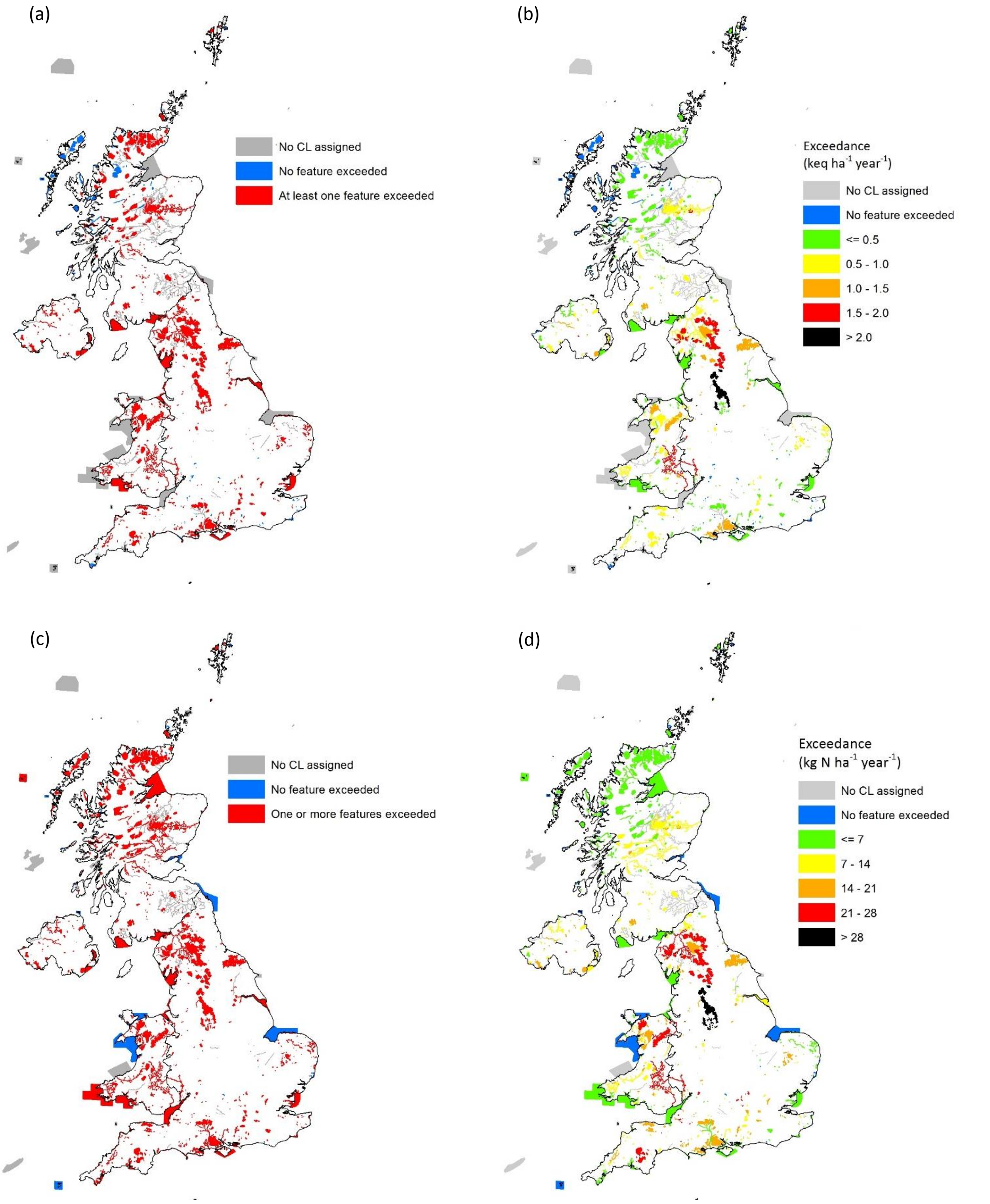


Figure 3.8: Critical load exceedance maps for SACs: (a) sites with exceedance of acidity critical loads for one or more features; (b) maximum Average Accumulated Exceedance (AAE) per site for acidity; (c) sites with exceedance of nutrient nitrogen critical loads for one or more features; (d) maximum Average Accumulated Exceedance (AAE) per site for nutrient nitrogen. All results based on site-relevant critical loads and CBED deposition for 2012-14.

3.3 Exceedance of critical levels

Concentration based critical levels have been defined for SO_2, NO_x and NH_3 (CLRTAP, 2004) and are summarised in Table 3.10.

Table 3.10: Critical levels for SO_2, NO_x and NH_3 (CLRTAP, 2015, chapter 3)

Pollutant	Vegetation type	Critical level µg m^{-3}	Time period
SO_2	Cyanobacterial lichens	10	Annual mean
	Forest ecosystems	20	Annual mean and half-year mean (Oct-Mar)
	(Semi-)natural	20	Annual mean and half-year mean (Oct-Mar)
NO_x (NO + NO_2 expressed as NO_2 µg m^{-3})	All	30	Annual mean
NH_3	Lichens & bryophytes##	1	Annual mean
	Higher plants###	3*	Annual mean

#including understorey vegetation

##including ecosystems where lichens & bryophytes are key part of the ecosystem integrity.

###including heathland, grassland and forest ground flora.

*An explicit uncertainty range of 2-4 µg m^{-3} was set for higher plants (including heathland, semi-natural grassland and forest ground flora). The uncertainty range is intended to be useful when applying the critical level in different assessment contexts (eg, precautionary approach or balance of evidence).

Python scripts have been written to generate the following critical level exceedance data:

(i) The UK land area (by country) with concentrations above the critical levels.

(ii) The broad habitat areas (by country) with concentrations above the critical levels; critical levels were not assigned to individual habitats. The habitat areas were based on the habitat distribution maps used for nutrient nitrogen critical loads.

(iii) The percentage of designated sites (SAC, SPA, SSSI) by country with concentrations above the critical levels; critical levels were not assigned to individual habitat features. Percentages were based on the number of sites where concentrations exceed the critical levels anywhere across a site.

All the exceedance results presented below for the UK land area, broad habitats and designated sites were based on these calculations and habitat/site data.

3.3.1 Exceedance of critical levels of SO_2

As described in Section 2.2 none of the UK CBED 5km data for SO_2 concentrations, based on the rural network, were above the critical levels. If the urban enhancement was included a few grid squares had values exceeding the critical level of 10 µg m^{-3}. The AEAT 1km UK SO_2 concentration data for 2009 showed more, though scattered, areas with concentrations between 10 and 26 µg m^{-3}. A comparison of the areas of the country exceeding the critical levels for SO_2 based on the urban enhanced 5km data and the AEAT 1km data (Table 3.11), showed that the results are an order of magnitude greater using the 1km data. However, in both cases the areas above the critical levels in the UK were very small (0.39% with the 1km data for the lowest critical level).

Table 3.11: Comparison of the land areas exceeding critical levels for SO_2 (10 and 20 μg m^{-3}) based on CBED 5km data from the rural network with an urban enhancement, and AEAT 1km data including the urban enhancement.

Country	Percentage area of land where concentrations exceed SO_2 critical levels based on:			
	CBED# 5km rural + urban enhancement		AEAT## 1km with urban enhancement	
	10 μg m^{-3}	20 μg m^{-3}	10 μg m^{-3}	20 μg m^{-3}
England	0.04	0	0.60	0.002
Wales	0	0	0.39	0.005
Scotland	0	0	0.04	0
NI	0.18	0	0.34	0
UK	0.03	0	0.39	0.001

#mean concentrations for 2010-12

##mean concentrations for 2009

3.3.2 Exceedance of critical levels of NO_x

As for SO_2, the CBED 5km NO_x concentration data based on the rural network alone did not exceed the critical level for NO_x (30 μg m^{-3}) anywhere in the UK. However, incorporating the urban enhancement into the CBED data (based on AEAT data) gave similar results to the AEAT 1km data that included the urban enhancement (Table 3.12), though the areas of the UK with exceedance of the critical level were very small (~3%) and primarily in urban areas.

Table 3.12: Comparison of the land areas exceeding critical level for NO_x (30 μg m^{-3}) based on CBED 5km data from the rural network with an urban enhancement, and AEAT 1km data including the urban enhancement.

Country	Percentage area of land where concentrations exceed NO_x critical level 30 μg m^{-3} based on:	
	CBED# 5km rural + urban enhancement	AEAT## 1km with urban enhancement
England	5.68	6.28
Wales	0.30	1.04
Scotland	0.16	0.21
NI	0.28	0.15
UK	3.13	3.53

#mean concentrations for 2010-12

##mean concentrations for 2009

3.3.3 Exceedance of critical levels of NH_3 based on CBED 5km concentration data

Exceedances of NH_3 critical levels were initially calculated using CBED 5km NH_3 concentration data and results based on data for 2005-2012 are presented below. More recently FRAME 1km NH_3 concentration data have been modelled (Section 2.2) and exceedances based on the different resolution concentration data sets are compared in Section 3.3.4 below and the latest results based on FRAME 1km data included in Section 3.3.5.

(a) UK land area exceeding critical levels of NH_3

NH_3 concentration data for the years 2005 to 2012 exceeded the critical level of 1 μg m^{-3} across 64-69% of the total land area of the UK, with little variation between the years (Table 3.13). The areas exceeding this critical level were largest for England and Northern Ireland (Table 3.13). The UK areas exceeding the critical level of 3 μg m^{-3} showed a general but small increase over the same time period; the percentage land area exceeding the critical level in Northern Ireland was much larger than in any other part of the UK, while there was hardly any exceedance of this threshold across Wales and Scotland (Table 3.14).

Table 3.13: Trends in the percentage land area with CBED 5km NH_3 concentrations exceeding the critical level of 1 µg m^{-3}.

Country	Land area (km^2)	Percentage area exceeded					
		2005-07	2006-08	2007-09	2008-10	2009-11	2010-12
England	131152	88.3	87.2	89.2	90.1	91.9	90.0
Wales	20761	57.6	54.9	59.4	60.9	65.5	60.9
Scotland	78744	21.8	21.8	23.7	24.3	26.1	24.2
NI	14177	90.6	90.5	91.8	92.3	93.4	92.1
UK	244834	64.4	63.6	65.7	66.6	68.6	66.5

Table 3.14: Trends in the percentage land area with CBED 5km NH_3 concentrations exceeding the critical level of 3 µg m^{-3}.

Country	Land area (km^2)	Percentage area exceeded					
		2005-07	2006-08	2007-09	2008-10	2009-11	2010-12
England	131152	8.6	8.1	10.9	12.1	16.2	12.0
Wales	20761	0.8	0.8	0.9	0.9	2.0	0.9
Scotland	78744	0.1	0.2	0.4	0.4	0.7	0.4
NI	14177	37.3	36.0	41.7	43.6	50.3	43.6
UK	244834	6.9	6.6	8.5	9.2	12.0	9.1

(b) Broad habitat areas exceeding the critical levels of NH_3

These calculations were based on the UK-scale habitat distribution maps generated for nutrient nitrogen critical loads work (Hall et al, 2015a). The results (Figures 3.9 and 3.10) reflected the spatial distributions of both the habitats and the NH_3 concentration data. The critical levels were not exceeded for montane habitat areas of the UK which are likely to be too high or too far from NH_3 sources. The distribution of Scots Pine occurs in areas with very low NH_3 concentrations; there are only 4 km^2 of this habitat where NH_3 concentrations were above 1 µg m^{-3}. The habitats with the largest areas with concentrations above the critical levels were calcareous grassland, managed broadleaf woodland, beech woodland and other unmanaged woodlands (Figures 3.9 and 3.10); these are all habitats that may be closer to NH_3 emission sources such as agricultural land or pig and poultry farms.

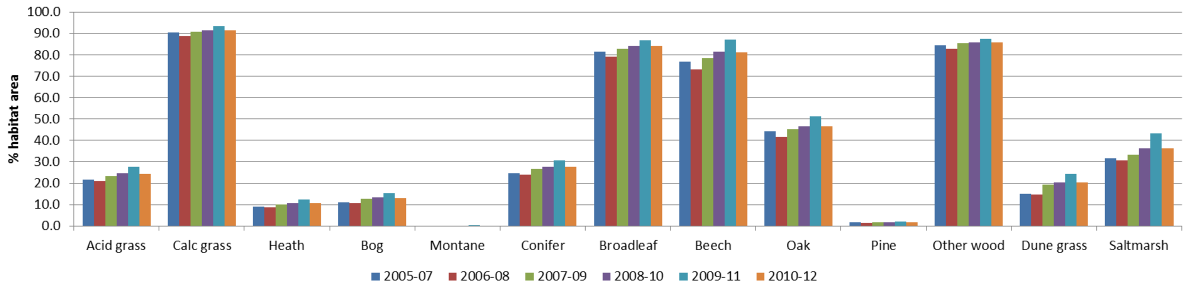


Figure 3.9: Percentage habitat area where NH_3 concentrations exceed 1 µg m^{-3}.

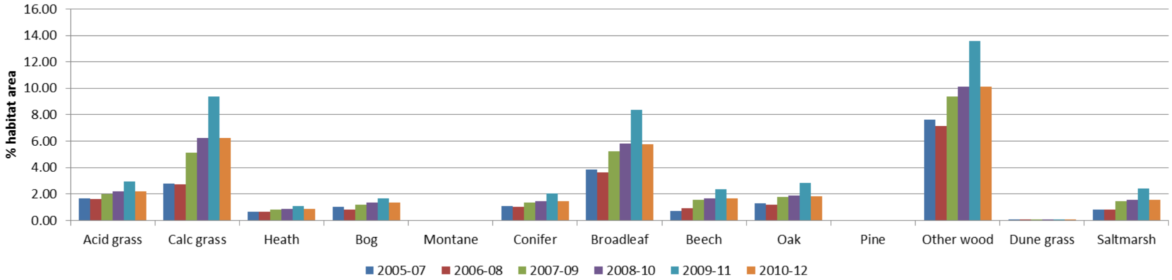


Figure 3.10: Percentage habitat area where NH_3 concentrations exceed 3 µg m^{-3}.

(c) Percentage of designated sites exceeding critical levels of NH_3

This analysis was only been carried out using the two most recent CBED datasets (2009-11 and 2010-12). The results for the UK are summarised in Table 3.15; they showed that more than half the sites are located in areas where the NH_3 concentrations exceed 1 µg m^{-3}, and around 10% where the concentration exceeds 3 µg m^{-3}.

Table 3.15: Percentage of designated sites in the UK where CBED 5km NH_3 concentrations for 2009-11 and 2010-12 (in brackets) exceed the NH_3 critical levels of 1 µg m^{-3} and 3 µg m^{-3} anywhere across a site.

Site type	Site count	Percentage of sites with concentrations for 2009-11 and 2010-12 (in brackets) that exceeding NH_3 critical levels :	
		1 µg m^{-3}	3 µg m^{-3}
SAC	615	68.0 (65.9)	13.2 (9.4)
SPA	255	61.6 (57.7)	11.4 (8.6)
SSSI	6869	77.3 (74.3)	10.6 (7.7)

3.3.4 Comparison of NH_3 critical level exceedances based on 5km and 1km NH_3 concentration data

This comparison was based on CBED 5km and FRAME 1km NH_3 concentration data for 2009-11 and 2010-12; the concentration maps for 2010-12 are shown in Figure 2.7.

(a) UK land area exceeding critical levels of NH_3

The total UK land area with NH_3 concentrations greater than the critical level of 1 µg m^{-3} was slightly higher for the 5km data than the 1km data (Table 3.16). Differences were more apparent for UK areas above 3 µg m^{-3}, reflecting the differences in resolution of the concentration data sets, with the higher concentrations, or "hotspots" being more clearly defined spatially when mapped using the 1km model, compared to the 5km data (eg, Figure 2.7). As a result, using the 1km data resulted in smaller areas exceeding the critical levels, and the 5km data potentially over-estimating the area exceeded (Table 3.17). The results for 2009-11 and 2010-12 showed the same general differences between using the 5km and 1km results, and also demonstrated that the NH_3 concentration data were similar for the two time periods.

Table 3.16: Comparison of the percentage land area with NH_3 concentrations above the critical level of 1 µg m^{-3} using 5km (CBED) and 1km (FRAME) NH_3 concentration data.

Country	Land area (km^2)	Percentage land area exceeded by:			
		5km 2009-11	1km 2009-11	5km 2010-12	1km 2010-12
England	131152	91.9	88.9	90.0	88.5
Wales	20761	65.5	58.9	60.9	59.0
Scotland	78744	26.1	19.7	24.2	19.8
NI	14177	93.4	85.6	92.1	86.0
UK	244834	68.6	63.9	66.5	63.7

Table 3.17: Comparison of the percentage land area with NH_3 concentrations above the critical level of 3 µg m^{-3} using 5km (CBED) and 1km (FRAME) NH_3 concentration data.

Country	Land area (km^2)	Percentage land area exceeded by:			
		5km 2009-11	1km 2009-11	5km 2010-12	1km 2010-12
England	131152	16.2	5.3	12.0	5.5
Wales	20761	2.0	0.9	0.9	0.9
Scotland	78744	0.7	0.1	0.4	0.2
NI	14177	50.3	13.4	43.6	14.7
UK	244834	12.0	3.7	9.1	3.9

(b) Broad habitat areas exceeding the critical levels of NH_3

As in (a) using the 1km NH_3 concentration data resulted in smaller areas exceeding the critical levels than using the 5km concentration data: 5% lower for the critical level of 1 µg m^{-3} and 1.6% lower for the critical level of 3 µg m^{-3}. The differences in the results for the 1km and 5km data for the exceedance of the 1 µg m^{-3} critical level were greatest for managed coniferous woodland, unmanaged beech woodland, bog and acid grassland; whereas the largest differences for the exceedance of the 3 µg m^{-3} critical level were for “other unmanaged woodland”, calcareous grassland and managed broadleaved woodland (Table 3.18). These differences reflected the spatial distributions of the habitats across the UK and the spatial patterns of the 5km and 1km NH_3 concentration data (Figure 2.7).

Table 3.18: Comparison of the percentage habitat areas with NH_3 concentrations for 2010-12 above the critical levels using 5km (CBED) and 1km (FRAME) NH_3 concentration data.

Broad Habitat	Habitat Area (km^2)	% area exceeding 1ug m^{-3}		% area exceeding 3ug/m^{-3}	
		5km NH_3 data	1km NH_3 data	5km NH_3 data	1km NH_3 data
Acid grassland	15235	24.4	18.2	2.22	0.58
Calcareous grassland	3578	91.5	88.1	6.25	1.77
Dwarf shrub heath	24826	10.6	6.5	0.88	0.21
Bog	5526	13.2	6.7	1.36	0.10
Montane	3129	0.0	0.0	0.00	0.00
Coniferous woodland (managed)	8383	27.6	18.7	1.48	0.37
Broadleaved woodland (managed)	7482	84.0	80.6	5.78	2.12
Fagus woodland (unmanaged)	719	81.1	72.9	1.66	0.51
Acidophilous oak (unmanaged)	1434	46.7	42.5	1.82	0.66
Scots Pine (unmanaged)	204	1.7	1.4	0.00	0.00
Other unmanaged woodland	1761	85.8	81.8	10.14	3.22
Dune grassland	323	20.3	14.5	0.01	0.11
Saltmarsh	427	36.2	31.5	1.56	0.38
All habitats	73027	30.0	25.0	2.24	0.64

(c) Percentage of designated sites exceeding critical levels of NH_3

Some designated sites include coastal areas that fall beyond the land area for which the 1km concentration data are mapped; as a consequence the 1km data do not cover as many of the designated sites as the 5km data, which due to the larger grid size extends in places beyond the coastline. This led to small differences in the number of sites included in the comparison of areas exceeding NH_3 critical levels (Table 3.16). Due to the fact that much of the country receives NH_3 concentrations above 1 µg m^{-3} (Figure 2.7), the results of exceedance of this threshold were similar for both the 1km and 5km data. The percentage of sites in areas where NH_3 concentrations exceed the critical level of 3 µg m^{-3} was much smaller with both the 1km and 5km data (Table 3.19), and using the 1km NH_3 concentration data led to between 1.7% and 4.3% fewer sites being exceeded compared to using the 5km data.

Table 3.19: Comparison of the percentage of designated sites with NH_3 concentrations for 2010-12 above the critical levels anywhere across a site using 5km (CBED) and 1km (FRAME) NH_3 concentration data.

Site Type	Site Count (where 5km NH_3 data exist)	Site Count (where 1km NH_3 data exist)	% sites exceeding 1ug m^{-3}		% sites exceeding 3ug/m^{-3}	
			5km NH_3 data	1km NH_3 data	5km NH_3 data	1km NH_3 data
SAC	615	614	65.9	62.4	9.4	5.1
SPA	255	245	57.7	54.3	8.6	6.9
SSSI	6869	6839	74.3	72.7	7.7	3.6

3.3.5 Exceedance of critical levels of NH_3 based on FRAME 1km concentration data

Section 3.3.4 compares using 1km and 5km ammonia concentration data for calculating exceedances of critical levels. However, as we recommend using the 1km FRAME concentration data (Section 2.2) this section presents a summary of the ammonia critical level exceedance results to date based on FRAME data for 2009-11, 2010-12 and 2011-13 (Tables 3.20, 3.21, 3.22). Overall there was little difference in the results for the different years, but they showed that ~63% of the UK land area receives ammonia concentrations above 1 µg m^{-3} and less than 4% receives concentrations greater than 3 µg m^{-3} (Table 3.20). Approximately 25% of the habitat areas mapped as sensitive to nitrogen (ie, based on the habitat distribution maps generated for nitrogen critical loads work; Hall et al, 2015a) were in areas with NH_3 concentrations above 1 µg m^{-3} and less than 1% in areas where concentrations were above 3 µg m^{-3} (Table 3.21). Approximately 60% of SACs in the UK were located in areas where the NH_3 concentrations exceed 1 µg m^{-3}, while less than 10% of UK SACs were in areas where the critical level of 3 µg m^{-3} was exceeded (Table 3.22).

Table 3.20: Percentage land area with ammonia concentrations exceeding critical levels of 1 and 3 µg m^{-3} using FRAME concentration data for 2009-11, 2010-12, 2011-13.

Country	Area (km^2)	Percentage land area with NH_3 concentrations > 1 µg m^{-3}			Percentage land area with NH_3 concentrations > 3 µg m^{-3}		
		2009-11	2010-12	2011-13	2009-11	2010-12	2011-13
England	131152	88.9	88.5	86.9	5.3	5.5	5.2
Wales	20761	58.9	59.0	57.6	0.9	0.9	0.7
Scotland	78744	19.7	19.8	19.4	0.1	0.2	0.2
NI	14177	85.6	86.0	62.7	13.4	14.7	15.6
UK	244834	63.9	63.7	62.7	3.7	3.9	3.8

Table 3.21: Percentage area of nitrogen-sensitive habitats in the UK receiving ammonia concentrations above the critical levels of 1 and 3 1 µg m^{-3}, based on FRAME concentration data for 2009-11, 2010-12, 2011-13.

Habitat	Area (km^2)	Percentage habitat area with NH_3 concentrations > 1 µg m^{-3}			Percentage habitat area with NH_3 concentrations > 3 µg m^{-3}		
		2009-11	2010-12	2011-13	2009-11	2010-12	2011-13
Acid grassland	15235	17.9	18.2	18.2	0.52	0.58	0.61
Calcareous grassland	3578	88.9	88.1	85.4	1.69	1.77	1.87
Dwarf shrub heath	24826	6.5	6.5	6.5	0.18	0.21	0.23
Bog	5526	6.6	6.7	6.9	0.08	0.10	0.10
Montane	3129	0.0	0.0	0.0	0.0	0.0	0.0
Managed conifer	8383	18.6	18.7	18.7	0.30	0.37	0.37
Managed broadleaf	7482	81.3	80.6	77.9	2.03	2.12	2.07
Beech woodland	719	75.9	72.9	66.1	0.56	0.51	0.39
Oak woodland	1434	43.8	42.5	39.2	0.64	0.66	0.67
Scots pine woodland	204	1.4	1.4	1.4	0.0	0.0	0.0
Other unmanaged wood	1761	82.0	81.8	79.9	3.02	3.22	3.23
Dune grassland	323	14.4	14.5	14.4	0.08	0.11	0.13
Saltmarsh	427	31.6	31.5	28.1	0.45	0.38	0.28
All habitats	73027	25.1	25.0	24.4	0.59	0.64	0.66

Table 3.22: Percentage of SACs with ammonia concentrations above the critical levels of 1 and 3 1 µg m^{-3} anywhere across a site, based on FRAME concentrations for 2009-11, 2010-12, 2011-13

Habitat	Site no.	Percentage of sites with NH_3 concentrations > 1 µg m^{-3}			Percentage of sites with NH_3 concentrations > 3 µg m^{-3}		
		2009-11	2010-12	2011-13	2009-11	2010-12	2011-13
England	231	94.4	93.5	90.0	10.0	10.0	13.4
Wales	85	72.9	75.3	72.9	0.0	1.2	4.7
Scotland	234	20.9	19.7	20.1	0.9	1.3	1.7
NI	54	85.2	87.0	87.0	3.7	3.7	5.6
England/Wales border	7	100	100	100	42.9	28.6	42.9
England/Scotland border	3	100	100	100	0.0	0.0	33.3
UK	614	62.7	62.4	60.9	4.9	5.1	7.5

4. Work Package 4: Model Inter-Comparison Project

Summary

- *Deposition data sets from different models (CBED, FRAME, EMEP4UK, NAME, CMAQ) were processed to convert all to the same grid system (ie. Ordnance Survey Great Britain Grid at 5km resolution). These data, together with calculated deposition budgets for the UK were provided to David Carslaw to complete work on the Model Inter-Comparison project (Defra contract AQ0936).*
- *A short report was submitted to Defra in July 2015 on the results of a preliminary comparison of ecosystem-specific deposition from CBED and FRAME and the associated critical load exceedance results. Further work is required to include EMEP4UK into this comparison.*

The following tasks have been completed under this contract as a contribution to AQ0936:

- Separate data sets of wet and dry deposition for NAME, CMAQ-HERTS, CMAQ-JEP were provided by David Carslaw (Kings College London) on a regular longitude/latitude grid, and converted to the Ordnance Survey Great Britain (OSGB) grid at 5km resolution.
- Separate wet and dry deposition data for CBED and all models (FRAME, EMEP4UK, NAME, CMAQ) were provided in comma delimited (.csv) format for 5km grid squares of the OSGB grid to David Carslaw.
- A short python script was written to calculate the ratios (as a percentage) of the difference between modelled (separate wet and dry) deposition and CBED deposition (as the reference data set). The results were exported to .csv format and provided to David Carslaw.
- A short python script was written to calculate the budget of wet and dry deposition for CBED and all models; results supplied by country in Excel format to David Carslaw.
- Results of the above included in the project (AQ0936) report to Defra, with the exception of the ratio data which do not appear to have been used.

It was agreed with Defra that the next steps (under this contract AQ0826) would be to make a comparison of the deposition and critical loads exceedances using grid-average deposition and ecosystem specific deposition values for 2010-12; the earlier results were based on grid-average deposition only as ecosystem-specific deposition was not available from all models. During 2015 grid-average and ecosystem specific deposition from the FRAME model (not calibrated to CBED) were compared with CBED data, and exceedances based on the different datasets also compared. The results were submitted in a separate report to Defra in July 2015 (Hall et al, 2015c). Preliminary calculations were also carried out using data from the EMEP4UK model, but these have not been included in the report as further work is required to ensure compatibility and consistency with moorland and woodland deposition from CBED and FRAME.

5. Work Package 5: Maintenance of critical loads and deposition websites

Summary

Two new project websites were created:

- *National critical loads and dynamic modelling: http://www.cldm.ceh.ac.uk*
  This was created to provide information on the role of the National Focal Centre; critical loads definitions, their calculations and use; exceedance metrics, maps and trends; publications and downloadable reports including the "Methods Report" and "Trends Report".
- *Pollutant deposition: http://www.pollutantdeposition.ceh.ac.uk*
  This was created to provide information about the pollutant monitoring networks, pollutant maps, and downloadable deposition datasets.

This work package relates to the following two websites:

(i) National critical loads and dynamic modelling: http://cldm.defra.gov.uk

(ii) Pollutant deposition: http://pollutant-deposition.defra.gov.uk

These websites were transferred to Defra templates and the Defra domain a few years ago (~2008/2009). However, the Defra website and content had since been reviewed and these science project websites were not moved to the new Defra site as it was agreed with Defra that CEH should host these websites instead. During 2014 and 2015 the CEH website also underwent a major update with a new website live from 23rd March 2015. Using the new CEH project templates, two new project websites were created and the content of the earlier websites transferred to the new websites on the CEH domain:

- National critical loads and dynamic modelling: http://www.cldm.ceh.ac.uk
- Pollutant deposition: http://www.pollutantdeposition.ceh.ac.uk

This enabled us as project managers to update the websites as required without going through a third party, ensuring the websites could be kept up to date and relevant. The websites provided a resource for a wide range of users (e.g. Defra, Devolved Administrations, SNCBs, Environment Agency, SEPA, researchers, other NFCs, etc) of the national critical load and exceedance data and the concentration and deposition data used for the calculations of critical load and critical level exceedances. They provided information to guide the user in the use and interpretation of the data in addition to results and/or data. The "CLDM" website provided, through the "UK Status Reports", a record of how critical loads have been developed in the UK from the late 1990's to the present day, with full details of the current methods for calculating critical loads and exceedances available in the downloadable "Methods Report", providing transparency of the methods and data used in the UK. Below is an overview of the information available from the two websites:

National critical loads and dynamic modelling: http://www.cldm.ceh.ac.uk

- Role of the National Focal Centre.
- Critical load definitions, their calculations, use and interpretation, uncertainties, site-specific critical loads, critical load maps and a summary of data available.
- Exceedances: calculation of exceedance metrics, trends in UK critical load exceedance summary statistics from 1995 to present day, exceedance maps.
- Introduction to dynamic modelling (to be expanded to include biodiversity-based critical loads).

- Publications: downloadable reports (e.g. Methods Report, Trends Report, UK Status Report), links to CLRTAP publications (e.g. UNECE Mapping Manual, CCE Status Reports) and lists of papers or other publications for further information on critical loads and their applications.

Pollutant deposition: http://www.pollutantdeposition.ceh.ac.uk

- Information about the pollutant monitoring networks (including UKEAP) across the UK.
- Description and downloadable CBED deposition data (annual and 3-year means).
- Description of the FRAME model and related publications.
- Pollutant maps (CBED deposition, FRAME deposition and concentrations, maps of the monitoring network data).
- Information on the EMEP supersites.

6. Work Package 6: Provision of the UK National Focal Centre (NFC)

Summary

- *The project provided UK representation at annual meetings of UNECE Task Force of the International Cooperative Programme on Modelling and Mapping (ICP M&M) and workshops of the Coordination Centre for Effects (CCE).*
- *The NFC submitted the following UK data in response to calls from the UNECE Working Group on Effects (WGE) and CCE:*
    - *2012-14 data call: Biodiversity metric (mean rescaled habitat suitability for locally-occurring positive indicator species) values for 18 sites, representing 9 different EUNIS habitat classes and two deposition scenarios.*
    - *2014-15 data call: (i) Habitat suitability data for 40 sites (26 acid grassland and 14 heathland); (ii) National 1km acidity and nutrient nitrogen critical loads for UK habitats sensitive to acidification and/or eutrophication; (iii) Nutrient nitrogen critical loads for designated features of SACs and SPAs.*
- *The MADOC-MultiMOVE-HQI model chain has been developed and applied to the acid-sensitive habitats (bogs, dry acid grasslands, wet heaths, dry heaths) of 354 SACs to demonstrate the methodology for deriving new biodiversity-based critical load functions in the UK that fully take into account the combined effects of sulphur and nitrogen on both acidification and eutrophication and the associated impacts on biodiversity. The overall form of most of these functions was satisfactory, in that they showed increases in overall habitat quality with decreases in both nitrogen and sulphur deposition. Preliminary results were presented at the joint meeting of the ICP M&M and CCE in April 2016.*

6.1 Attendance at CCE Workshops and ICP Modelling and Mapping Task Force meetings

Jane Hall acted as the Head of the UK NFC and represented the UK at annual CCE Workshops and ICP Modelling and Mapping (ICP M&M) Task Force meetings. Ed Rowe and/or Chris Evans also attended these meetings as UK experts in dynamic modelling and biodiversity-based critical loads modelling. These meetings provided a forum for countries to present and discuss their progress in developing the methods and results needed for integrated assessment modelling activities under CLRTAP, and to plan future activities required for the CLRTAP Working Group on Effects (WGE) work plans, including "Calls for data" (see 6.2 below).

The following presentations have been given at the meetings over the last four years:

(a) 2013

Ed Rowe: Biodiversity Indicators for UK habitats.

(b) 2014

Jane Hall: Ammonia deposition reductions required to protect Habitats Directive Annex I habitats (see Section 11).

Ed Rowe: Selecting a biodiversity metric for the UK response to the CCE Call for Data by comparison with specialist judgement.

Chris Evans: A new approach for calculating acidity critical loads for peat soils (see Section 9).

(c) 2015

Ed Rowe: Deriving N and S critical load functions from thresholds set using empirical critical loads.

(d) 2016

Ed Rowe: Critical load functions: effects of uncertainties in biogeochemical and species responses and species choice.

All presentations were well received and contributed to the European discussions on the impacts of air pollution, in particular nitrogen, and the future development of critical loads, especially the methods aimed at protecting biodiversity.

In 2014 Ed Rowe presented to the CCE and ICP M&M a biodiversity metric, as an index of habitat quality (HQI) and defined as the mean habitat-suitability for positive indicator species. The HQI was defined in consultation with UK habitat specialists from the Statutory Nature Conservation Bodies (SNCBs) under Defra contract AQ0832. This approach was adopted at the 2014 CCE Workshop and Task Force meeting as a common standard for use by all parties to the Convention in the development of biodiversity-based critical load functions. The development and application of "biodiversity-based" critical load functions based on this metric is described in Section 6.3.

The minutes of the joint meetings of the CCE Workshop and Task Force of ICP M&M are reported to the Working Group on Effects annual meeting in September and subsequently published on the ICP M&M website (http://icpmapping.org/Activities), along with copies of the presentations made.

6.2 Responding to "Calls for data"

The NFC has responded to "Calls for Data" issued by the CCE in agreement with the Working Group on Effects. The calls under the contract to date are summarised below.

6.2.1 CCE Call for Data 2012-14

This Call was announced in November 2012 following the Working Group on Effects meeting in September 2012. The objective of the Call was for each country to compile output variables of soil-vegetation models for different EUNIS habitat classes to enable the calculation of country-specific biodiversity indicators for scenario assessment of changes in biodiversity on a regional scale. The final goal of the Call was to derive a harmonized metric from the submitted variables and indicators to quantify "no net loss of biodiversity" and make comparisons between regions and countries. To provide countries with sufficient time to develop their research and address this complex task the deadline of this Call was 3rd March 2014.

In the UK, the modelling work required for this Call was funded through two separate short (3-month) Defra contracts during 2013-14. The UK NFC submitted values for the biodiversity metric (mean rescaled habitat suitability for locally-occurring positive indicator species) values for 18 sites (two each for 9 different EUNIS habitat classes) and two deposition scenarios. Further details can be found in Hall et al (2014a).

6.2.2 CCE Call for Data 2014-15

This Call was for new "biodiversity-based" critical load functions for nitrogen and sulphur which define the maximum combined pollutant loads that prevent biodiversity declining below a critical threshold. The UK NFC submitted data for 40 sites (26 acid grassland and 14 heathland) in response to this call, using the MADOC-MultiMOVE dynamic model chain to predict soil and vegetation change and consequent changes in habitat suitability for plant and lichen species, using the HQI metric. The methods and results have been reported separately to Defra in Rowe et al (2015). The NFC also submitted:

(i) The national 1km acidity and nutrient nitrogen critical loads (based on empirical or mass balance methods) for broad habitats sensitive to acidification and/or eutrophication, in the updated database format required by the CCE. The critical loads data for the 1752 freshwater catchments were sub-divided to the same grid resolution for consistency with the terrestrial data and for compatibility with the new EMEP grid resolution; this approach was discussed and agreed with Chris Curtis (ENSIS). The critical loads themselves remained unchanged from previous versions, with the exception of some nutrient nitrogen critical loads that were for habitat areas also occupied by designated sites (see (ii) below).

(ii) Nutrient nitrogen critical loads for designated features of Special Areas of Conservation (SACs) and Specially Protected Areas (SPAs). In previous years the NFC had submitted critical loads for the designated features of SACs and SPAs (Hall et al, 2011b), however, as these sites can overlap with the UK broad habitat critical loads data they could not be used by the CCE due to double counting of habitat areas in integrated assessment modelling activities. To overcome this, the nutrient nitrogen critical loads for broad habitats, SACs and SPAs have been integrated into a single database, without duplicating the areas. This was achieved by:

    - Identifying the designated features that were the same EUNIS class as the UK broad habitats.
    - Identifying the 1km squares that contained individual UK broad habitats and all or part of any SAC and/or SPA.
    - Assigning the appropriate nutrient nitrogen critical load for each relevant EUNIS class to each 1km square, using the lowest value if there were differences between the values for the broad habitat, the SAC and/or SPA.
    - Assuming that the habitat area for the designated feature habitat within the 1km square was the same as the area that had been mapped for that broad habitat. This was necessary as spatial data on the location and areas of designated feature habitats within sites was not available.
    - Setting the "protection" score for the 1km squares according to the codes provided by the CCE (1: SPA, 2: SAC, 3: SPA and SAC, -1: protection status unknown).

    The critical load values applied to feature habitats of UK SACs and SPAs were values (within the published ranges) agreed nationally for use in air pollution impact assessments. For some habitats these values were the same as the "UK mapping values" applied to broad habitats and based on UK evidence; where no UK evidence exists, the values were based on expert opinion or set to the minimum of the published range. It should be noted that the resulting database tables submitted to the CCE did not include: (a) designated feature habitats that were not mapped nationally; (b) areas of SACs/SPAs that fell outside of the broad habitat areas mapped nationally. In total 13.3% of the UK 1km critical load records submitted for nutrient nitrogen represent the designated feature habitats of SACs and/or SPAs

The data submitted, and methods used, were documented in the UK chapter of the 2015 CCE Status Report (Hall et al, 2015d).

6.2.3 Call for Data 2015-17

This latest Call was issued in November 2015 with a delivery date in 2017 to allow NFCs sufficient time to develop their data and methods to respond to the Call. The specific aims of the Call were to:

(a) Derive nitrogen and sulphur critical load functions taking into account their impact on biodiversity (i.e, critical loads for biodiversity).
(b) Present plans and preliminary results on developing critical loads for biodiversity at the ICP M&M meeting in Dessau in April 2016.
(c) Offer NFCs the possibility to update their national critical loads data on acidity and eutrophication.
The results of this Call will be considered as an update of the European critical loads database, to be approved under CLRTAP for use by its bodies, and for possible use for European policy support. The CCE background database will be used for countries who do not respond to this Call for Data.

The UK work developed to date to enable the UK to deliver data in response to this Call is described in Section 6.3 below; preliminary results were presented at the ICP M&M meeting in April 2016.

6.3 Developing biodiversity-based critical load functions for the UK

This section introduces the concept of biodiversity-based critical load functions, the method and model development to date, and results of the first application of the approach to acid-sensitive habitats within UK SACs. Most of the biodiversity critical load functions showed a decline in the overall habitat quality as reflected by the HQI in response to both sulphur and nitrogen deposition. At this stage there are still uncertainties in the model chain and future work will aim to reduce these and increase the confidence in the model outputs.

6.3.1 Introduction to biodiversity-based critical load functions

Despite substantial reductions across the UK in sulphur (S) deposition and moderate reductions in total reactive nitrogen (N) pollution since peak levels in the late 20th century, many ecosystems remain in unfavourable condition due to air pollution. The total load of acidifying pollutants has decreased considerably, and there is evidence for widespread recovery of soil pH (Reynolds et al., 2013), although in some areas where soils are weakly buffered by cation weathering recovery from acidification is not yet evident (Evans et al., 2014). However, N is a eutrophying as well as an acidifying pollutant, and affects biodiversity by favouring fast-growing, competitive plant species at the expense of slower-growing, smaller species (Hodgson et al., 2014). Pollutant N also has direct toxic effects (Cape et al., 2009). In combination, these effects have caused substantial losses of plant and lichen species, in particular in areas of the southern UK and/or in higher-rainfall areas, where N pollution is greatest (Henrys et al., 2011; Stevens et al., 2012). There are knock-on effects on other aspects of biodiversity, for example through the loss of animal species that depend on threatened plant species or on warm microsites that tend to be lost when plant growth and litter production increase (Wallis de Vries and Van Swaay, 2006). For these reasons, N pollution has been identified as a major cause of biodiversity loss globally (Sala et al., 2000) and it is increasingly recognised that controlling N pollution is essential if biodiversity protection targets are to be met (Secretariat of the Convention on Biological Diversity, 2011).

Efforts under the CLRTAP to control acidifying pollutants makes use of the critical load concept. With increasing recognition of the significance of eutrophication as well as acidification, it is important that CL functions represent the combined effects of N and S pollution and fully take both acidification and eutrophication effects into account. It has therefore been necessary to move beyond a simple understanding of impacts on ecosystem chemistry, such as on soil pH, to take into account overall effects on biodiversity. The Coordination Centre for Effects (CCE) of the CLRTAP has therefore encouraged the development of dynamic modelling approaches that capture the combined effects of air pollution, and of metrics that summarise impacts on biodiversity (Hettelingh et al., 2008).

In previous Defra projects the MADOC biogeochemical model has been linked to the MultiMOVE species model, a biodiversity metric defined for use in this context, and methods developed for deriving biodiversity-based CL functions (Rowe et al. 2014a, 2014b and 2015). In response to the current CCE Call for Data (2015-17) we aim to provide in early 2017 a more geographically complete submission of CL_{biodiv} functions, which requires the approach to be upscaled. This section describes progress towards this goal made in the current project, on methods for linking MADOC to MultiMOVE, and on methods for upscaling to many sites. Model development for upscaling has focused on streamlining the code, and is illustrated by the application of the model chain to multiple Natura 200 sites.

A critical part of the model chain is the link between the abiotic variables output by MADOC and the Ellenberg score input variables required by MultiMOVE. There is a need to understand and quantify the relationships between Ellenberg scores and measured abiotic variables and understand the uncertainty involved in this step. The relationships are not necessarily straightforward as Ellenberg indicators were not developed to be a linear representation of particular abiotic variables (Ellenberg et al., 1991). Therefore there is the potential for complex non-linear relationships to occur between Ellenberg scores and measured variables. A number of previous studies have sought to construct calibration or linking equations between Ellenberg indicators and abiotic variables. Several large studies have constructed equations for the Netherlands flora due to the large amount of data present for this region (Ertsen et al., 1998; Schaffers and Sykora, 2000; van Dobben et al., 2006; Wamelink and van Dobben, 2003). Additional studies have focused on constructing equations for other countries or regions (Lawesson et al., 2003) (Andrianarisoa et al., 2009). Constructing equations for separate regions is considered sensible due to regional variation in flora and, in some cases, locally adjusted Ellenberg scores (Hill et al., 2000). For the UK several attempts at constructing calibration models have taken place, in particular for the key axis Ellenberg 'N' axis that is related to eutrophication (Rowe et al., 2011a; Rowe et al., 2014d; Smart et al., 2010). We have built on this work by quadrupling the number of plots included and extending the coverage, particularly to calcareous grasslands.

Almost all previous work linking Ellenberg indicators to abiotic variables has used linear-regression-based or correlative methods. Non-linear relationships have been considered in some papers (Ertsen et al., 1998) (Lawesson et al., 2003) (van Dobben et al., 2006) (Seidling and Fischer, 2008) but not all (Andrianarisoa et al., 2009; Schaffers and Sykora, 2000; Wamelink and van Dobben, 2003) even in cases when visual inspection of the data suggests a non-linear relationship (Schaffers and Sykora, 2000; Wamelink and van Dobben, 2003). Very few studies have considered the potential for multiple predictors of Ellenberg scores, generally focusing on one-to-one correspondence with measured variables. In cases where multiple predictors were considered (Lawesson et al., 2003), models with more than one predictor were found to have better predictive ability than single term models. No studies appear to have considered interactions between predictors (e.g. that the effect of soil nitrogen on EbN may be more or less at a given pH). However, including interaction terms complicates interpretation of Ellenberg values so it may be considered sensible not to include them. The aim of this aspect of the work is both to better understand the quantitative links between community mean Ellenberg scores and measured abiotic variables, and to produce predictive equations for the UK flora which can be utilised in the MADOC-MultiMOVE-HQI model chain. Parameters for existing equations were updated with new data and new equations developed using a range of techniques. The predictive power of single and multiple predictor linear regression models was investigated.

6.3.2 Methods for biodiversity-based critical load functions

Step 1: Biogeochemistry – Species – Habitat quality modelling

The methods used to calculate CL_{biodiv} functions in the current project have not changed substantially since those developed in the AQ0840 project (Rowe et al., 2015) and used for the previous Call for Data. These methods are summarised here and in Figure 6.1. Essentially, effects of N and S pollution on soil and vegetation biogeochemistry are simulated using the MADOC dynamic model (Rowe et al., 2014e) which predicts changes in soil pH, total C/N ratio and plant-available N, and in vegetation canopy height. The MADOC model is calibrated to the values of soil pH and total C/N ratio that were observed for the habitat in the Countryside Survey (Emmett et al., 2010), by adjusting calcium weathering rate or the density of exchangeable protons on dissolved organic carbon to match pH, and by adjusting the rate of N fixation during the pre-industrial period to match C/N.

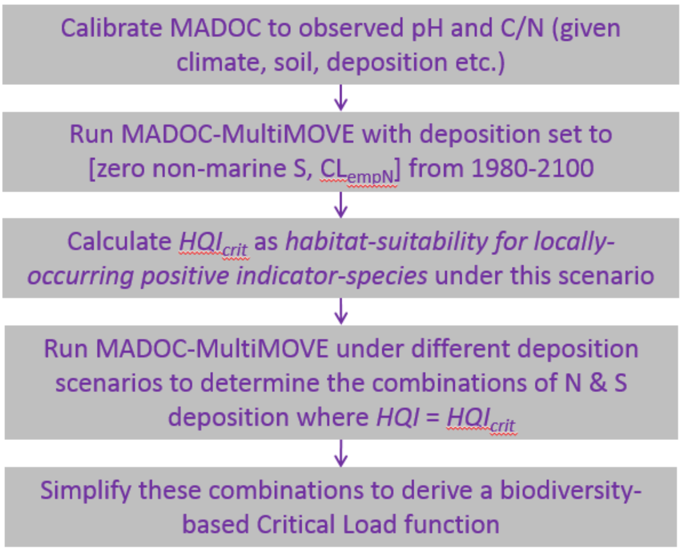


Figure 6.1: Summary of workflow used to generate biodiversity-based critical load functions using the MADOC-MultiMOVE model chain.

The effects of simulated conditions on habitat suitability for a set of plant species are simulated using MultiMOVE (Henrys et al., 2015) and corrected for prevalence in the training data using the method of Real *et al.* (2006). The MADOC outputs are abiotic parameters such as soil pH, whereas MultiMOVE uses trait-means such as mean 'Ellenberg N' score to characterise the environment. To convert from abiotic conditions to trait-means a set of transfer functions are used, which have been derived from empirical data. More data were collated within the current project for fitting such functions, but this fitting was not completed in time for use in the simulations presented here. The data presented in this report use the same functions as were used in Rowe et al. (2015), which for completeness are shown in Table 6.1.

A site-specific threshold value of *HQI* below which the site is considered to be damaged or in unfavourable condition, HQI_{crit}, is calculated as the *HQI* value obtained after running the MADOC-MultiMOVE model chain with N deposition set at the empirical CL for N, with zero non-marine S deposition, for the period 1980-2100. This makes use of the results from the major review of evidence that was used to establish empirical critical loads for N (Bobbink and Hettelingh, 2011).

Once the HQI_{crit} value has been established for the site, the model chain is then run with 100 different combinations of N and S deposition, covering the range from 0-180 % of CL_{empN} and 0-180 % of CL_{maxS}. The resulting values of *HQI* can be plotted as a response surface (shown for a hypothetical site in Figure 6.2a). The contour on this surface where $HQI = HQI_{crit}$ represents the maximum combinations of N and S deposition that are compatible with protecting the site from damage. For the response to the current Call for Data (CCE, 2014), this contour needs to be simplified into a form that can be represented by two points on this plane, [CL_{Nmin}, CL_{Smax}] and [CL_{Nmax}, CL_{Smin}], as shown in Figure 6.2b. These two points were positioned by minimising the total sum of squared differences between the simplified function and the HQI_{crit} contour.

Table 6.1: Conversion equations used to estimate floristic trait-means (used to predict habitat-suitability for species) from biogeochemical conditions. E_W = mean Ellenberg 'moisture' score for species present; E_R = mean Ellenberg 'alkalinity' score for present species; E_N = mean Ellenberg 'fertility' score for present species; G_H = mean Grime 'height' score for present species; MC = soil moisture content, g water 100 g^{-1} fresh soil; pH = soil pH; N_{av} = available N, g N m^{-2} yr^{-1}; *CN* = CN ratio, g C g^{-1} N; *H* = canopy height, cm; C_{plant} = total plant biomass C. Mean G_H was weighted by observed cover or occurrence frequency; other trait-means were not weighted.

Value to be estimated	**Calculated as**	**Source**
E_W	$$\frac{\ln\left(\frac{MC}{100-MC}\right)+3.27}{0.55}$$	Smart et al. (2010)
E_R	$$\frac{pH-2.5}{0.61}$$	Smart et al. (2004)
E_N	$$0.318\log_{10}N_{av}+1.689+\frac{284}{CN}$$	Rowe et al. (2011b)
G_H	$$max(1, 1.17\times\ln H-1.22)$$	Rowe et al. (2011b)
H	$$\left(\frac{C_{plant}}{14.21\times 3}\right)^{1/0.814}$$	derived from Parton (1978) and Yu et al. (2010)

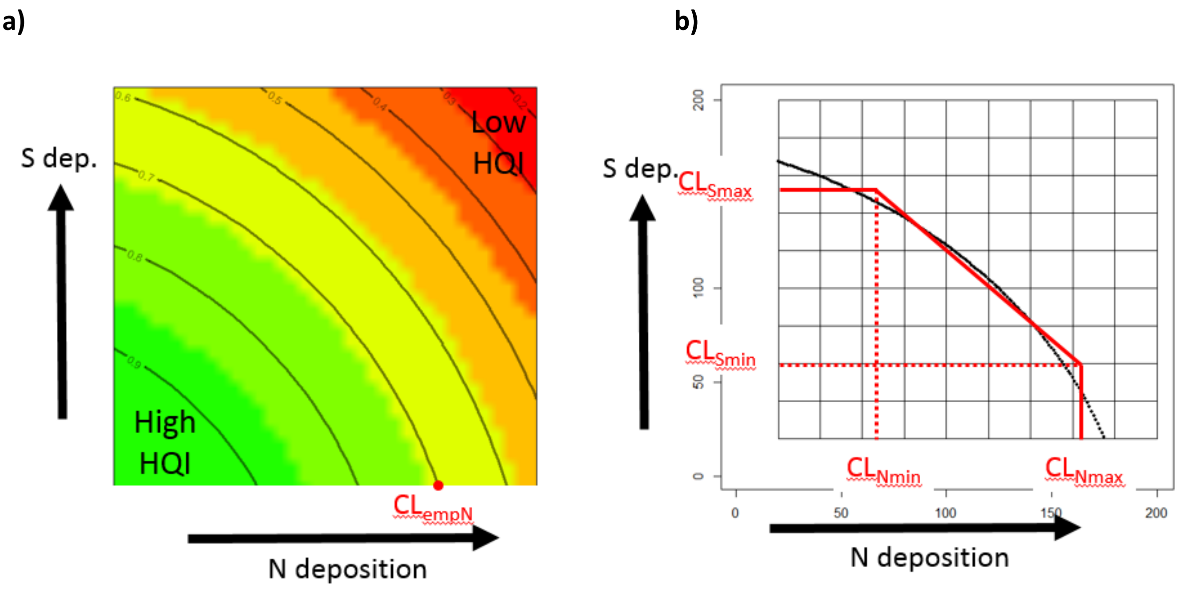


Figure 6.2: a) Response to combined loads of nitrogen and sulphur of a habitat quality index (*HQI*), as shown by colours and contours, at a hypothetical site; b) contour (black line) where *HQI* reaches a critical value, corresponding to a 'biodiversity-based' Critical Load function, and simplified version of this function by two points (red line).

Step 2: Site selection

A total of 415 UK Special Areas for Conservation (SACs) were identified from the SRCL database as having one or more acid-sensitive Annex I habitats as a designated feature (

Table 6.2). Within each of these SACs, 1km grid squares that have the corresponding EUNIS habitat (according to national-scale habitat distribution maps generated for UK critical loads work (Hall et al, 2015)) were identified, and one of these was chosen at random for the model application. UK-wide digital data on the location of designated feature habitats within SACs is not available, so the location of the grid square chosen does not necessarily correspond to the location of the designated feature.

Data for setting up MADOC were obtained from the NFC critical loads database, and local species-lists for filtering MultiMOVE outputs were obtained from the Biological Records Centre. The model chain was run for each of these squares, although CL_{biodiv} functions were only successfully generated for 82-90% of SACs with each habitat, due to some missing values in the underlying data, resulting in 354 calculated CL_{biodiv} functions (

Table 6.2).

Table 6.2: Numbers of Special Areas for Conservation (SACs) with different acid-sensitive habitats for which calculation of biodiversity-based Critical Load (CL_{biodiv}) functions was a) attempted; and b) successful.

EUNIS class	EUNIS class name	Annex 1 habitat codes*	Short name	a) Number of SACs	b) CL_{biodiv} calculated
D1	Raised and blanket bogs	H7110 H7120 H7130	Bogs	126	103
E1.7	Closed non-Mediterranean dry acid and neutral grassland	H6230	Dry acid grassland	43	39
F4.11	Northern wet heaths	F4010 F4020	Wet heath	119	107
F4.2	Dry heaths	F4030 F4040	Dry heath	127	106
			Total	**415**	**355**

*Annex 1 habitat codes related to the EUNIS classes assigned to SAC features in the SRCL database.

Step 3: Assessing uncertainty in transfer functions

Eight datasets were collated to build and test the transfer functions (Table 6.3). In each dataset the full vegetation community was recorded from quadrats with co-located soil measurements. The soil measurements made varied between datasets. For each dataset, unweighted mean community Ellenberg indicators were calculated for Ellenberg N, F and R. Only vascular plants were used to calculate mean Ellenbergs as bryophytes were not recorded in all surveys.

Table 6.3: Datasets with co-located soil measurements and floristic records used to assess transfer functions between abiotic measurements and trait-means.

Dataset	Habitat	Geographical coverage	Number of plots	Soil measures used in this study
(Rowe et al., 2016)	Bracken	Wales and northern England	49	pH, GWC, VWC, total N, total C, CN ratio
(Carey et al., 2008)	Multiple	England, Scotland and Wales	3308	pH, GWC, VWC, total N, total C, LOI, Olsen P, CN ratio, mineralisable N
(Emmett et al., 2016)	Multiple	Conwy catchment in Wales	35	pH, total N
(Stevens et al., 2010)	Grasslands	England, Scotland and Wales	74	pH, total N, total C, CN ratio
(Stevens et al., 2015)	Grasslands	Peak District	453	pH
(Emmett et al., 2014)	Multiple	Wales	277	pH, GWC, VWC, total N, total C, CN ratio, LOI, Olsen P
(Rich et al., 2015)	Calcareous grasslands	England, Scotland and Wales	48	pH
(Critchley et al., 2002)	Grasslands	England	243	pH, CN ratio

Two approaches were used to fit models between Ellenberg means and soil data: single-predictor explanatory models; and multiple-predictor explanatory models.

a) Single-predictor explanatory models.

Models were fitted with a single abiotic variable as a predictor to assess the ability for any of the measured variables to explain a large proportion of variance in Ellenberg scores. If found to be the case, this would not only simplify the interpretation of Ellenberg scores but also the equations needed to transfer between Ellenberg scores and abiotics in dynamic model chains. Each of the studied Ellenberg indicators (N, R and F) were modelled with a single predictor in a linear mixed model whereby Site nested in Survey were included as random intercepts. For Ellenberg R, which represents 'reactivity' or pH preference, soil pH was the only predictor for which there was a theoretical basis to construct a model, so only this single predictor was tested. Because the Ellenberg R-pH relationship was clearly non-linear in this dataset (Figure 6.1a), both quadratic and breakpoint models were tested. For Ellenbergs F and N, multiple soil variables were measured which could feasibly be related to the Ellenberg score in a single predictor model. For Ellenberg F, both gravimetric and volumetric soil water contents could be derived for some datasets. It might be predicted that volumetric water content would be a better predictor of Ellenberg F because it accounts for variation in bulk density between soil types and should therefore be a better representation of soil moisture. However, bulk density is not always measured so gravimetric water content might be a more practical predictor. To test which measurement was the best predictor of Ellenberg F, two models were constructed, one with each predictor and the best model chosen using AIC. The amount of data available to fit the two models was different between the two predictors because bulk density was not always available to derive volumetric water content, therefore to compare models using AIC the dataset was reduced to that with complete cases of gravimetric and volumetric water content. The best model was then fitted with the full dataset to derive the terms for the equation.

A larger number of predictors could be expected to potentially predict Ellenberg N, an index of productivity or fertility. Total carbon, nitrogen, and loss on ignition are all commonly measured characteristics of soil potentially linked to fertility and all are highly correlated with each other so relatively similar performance might be expected from models built with each term as the single predictor. Carbon to nitrogen ratio can also be derived from this data and has been used in models to predict Ellenberg N previously. Phosphorus availability is also a likely predictor of productivity and is most often measured using the Olsen methodology. Although this method has been criticised in previous studies (DeLuca et al., 2015) it is commonly the only P measure available in existing datasets so the ability of Olsen P to predict Ellenberg N was tested here. As with Ellenberg F, the models were fitted with the complete dataset (all variables measured) and the best single predictor model was chosen using AIC. The best model was then fitted with the full dataset to derive the terms for the equation.

b) Multiple-predictor explanatory models

Ellenberg indicators were also modelled using multiple predictors to investigate the degree to which adding additional predictors increased model performance. Each Ellenberg indicator modelled against a "complete" set of predictors (pH, total nitrogen, CN ratio, Olsen P and volumetric water content). Both total carbon and gravimetric water content were highly collinear with other predictors and were therefore not included in the full model. A decision was made to fit the same set of predictors to each response to avoid pre-selecting predictors for each Ellenberg indicator based on potentially faulty assumptions. Quadratic terms for each predictor were also included in the full model. If any terms in the full model were considered non-significant, the terms were removed and the new model compared with the old model using AIC. If the difference in AIC was more than 2 after removing the model term then the new model was kept as the best model. Models were reduced stepwise in this fashion, with quadratic terms removed first. Not all datasets in Table 1 recorded all predictors, therefore model selection was based on only the plots where all variables were recorded. Final models were fitted on the full dataset (although omitting any rows with missing data and therefore defaulting to the model selection dataset if all terms were retained).

Both single and multiple term models were evaluated using the Nakagawa R^2 implemented in the MuMIn package in R (Barton, 2016) which calculates marginal and conditional R^2 values which can be equated to the predictive power of the model without and with information from the random effects respectively.

The two approaches outlined above seek to find the best models for describing the relationship between Ellenberg N and soil variables with the aim of understanding Ellenberg-soil relationships. The models are all directional, with the Ellenberg score as the response variable. However, it is sometimes necessary to reverse the models to predict soil variables from Ellenberg scores. For example, calibrated dynamic models such as the MADOC model require an initial calibration step whereby model parameters are adjusted slightly to match the input data. For this it is important that the equations used to transfer between Ellenberg and soil variables are completely reversible. A linear model framework is not ideal for this for several reasons. Firstly, model fitting assumes that predictors (i.e. soil variables) are measured without error, an assumption that does not hold if the model is reversed. Secondly, when Ellenberg models have multiple predictors it is difficult to reverse them in a way that makes sense. Thirdly, non-linear terms such as quadratic functions have two solutions when inverted. Therefore, an alternative approach is needed to construct equations which can be used in model calibration.

6.3.3 Results for biodiversity-based critical load functions

Deriving a biodiversity-based critical load function for a single site

The outputs of MADOC-MultiMOVE-HQI are illustrated for an example dry acid grassland ("Hill of Towanreef" SAC in Aberdeenshire) in Figure 6.3. The value of *HQI* was calculated under scenarios in which combinations of N deposition (0-180% of CL_{empN}) and S deposition (0-180% of CL_{maxS}) were applied between 1980 and 2100, and plotted as a response surface (Figure 6.3a). This shows that at low levels of N and S deposition, habitat quality as expressed using HQI is relatively high. The HQI value declines with increasing loads of either S or N, and (at least initially) this decline is more rapid with increasing N (Figure 6.3a). This response surface is illustrated on a simple plane in Figure 6.3b, which shows the combinations of N and S deposition that result in a site-specific critical value, HQI_{crit} (0.324 in this case). The HQI_{crit} value was determined by simulating a scenario with N deposition set at CL_{empN} and zero non-marine S deposition, so while the shape of this function may vary, in all cases it meets the x axis at this point. The “Call for Data 2015-17” requires only a simplified version of this function for each site, described by two points on this plane: [CL_{Nmin}, CL_{Smax}] and [CL_{Nmax}, CL_{Smin}]. The simple function is interpolated between and extrapolated beyond these points, as shown by the red line in Figure 6.3c. In the case of this site, CL_{empN} = 714 eq ha^{-1} yr^{-1} and CL_{maxS} = 1550 eq ha^{-1} yr^{-1}, so the fitted values are:

CL_{Nmin} = 26 % of CL_{empN}	= 183 eq ha^{-1} yr^{-1}	= 2.6 kg N ha^{-1} yr^{-1}
CL_{Smax} = 172 % of CL_{maxS}	= 2661 eq ha^{-1} yr^{-1}	= 42.7 kg S ha^{-1} yr^{-1}
CL_{Nmax} = 95 % of CL_{empN}	= 678 eq ha^{-1} yr^{-1}	= 9.5 kg N ha^{-1} yr^{-1}
CL_{Smin} = 1 % of CL_{maxS}	= 16 eq ha^{-1} yr^{-1}	= 0.2 kg S ha^{-1} yr^{-1}

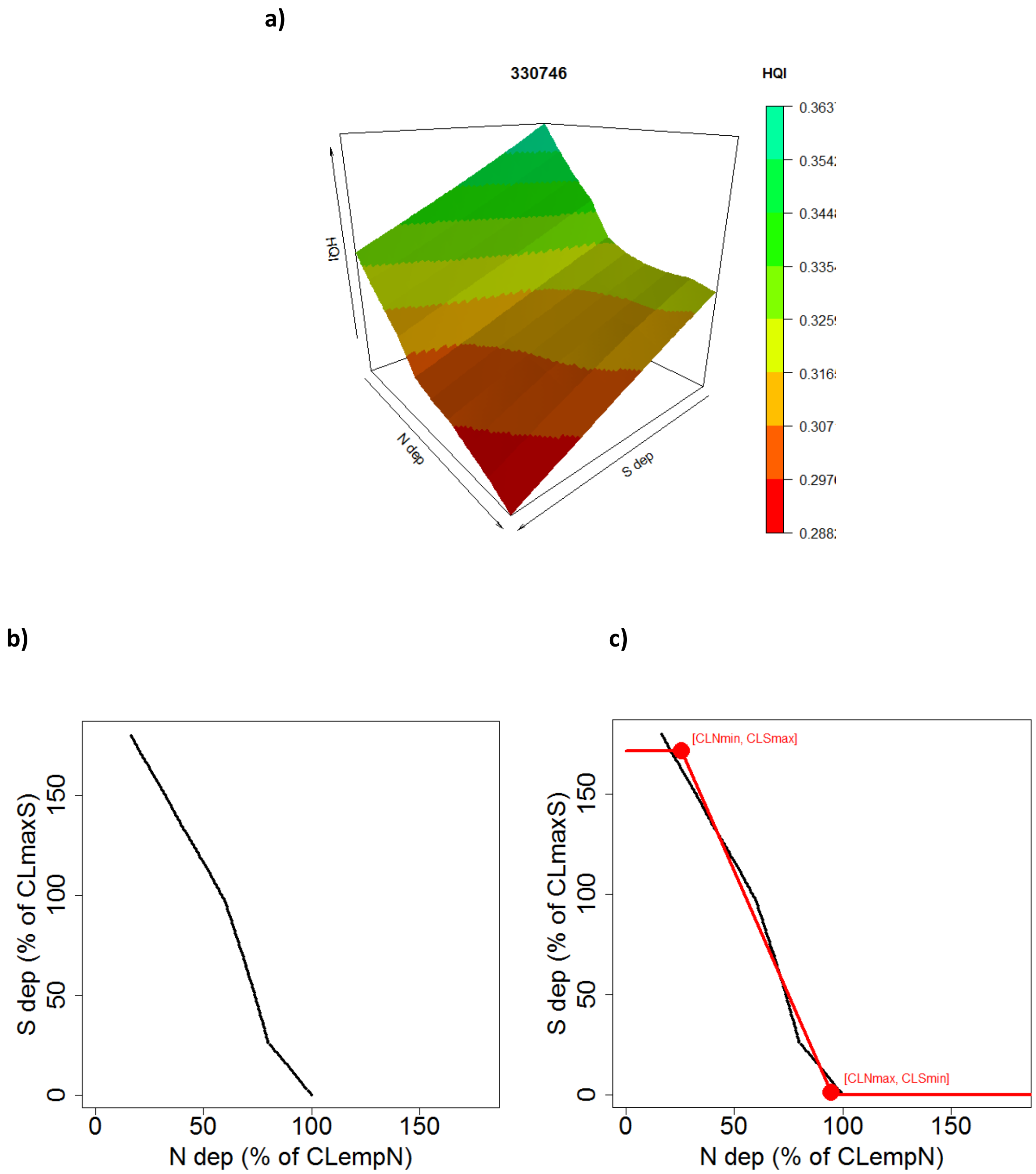


Figure 6.3: Response of habitat quality of an acid grassland Special Area for Conservation, Hill of Towanreef, to nitrogen (N) and sulphur (S) deposition scenarios: a) response illustrated in a ‘3D’ plot; b) combinations of N and S deposition resulting in a critical value for habitat quality, HQI_{crit}; c) simplified version of the HQI_{crit} response function.

Biodiversity-based critical load functions for UK acid-sensitive SACs

The MADOC-MultiMOVE model chain was used to generate response surfaces and simplified biodiversity-based CL functions for 354 sites. As noted above, the CL_{Nmax} value is always close to the empirical CL for N, since the latter is used to define the critical threshold for *HQI*. The main variation in the CL_{biodiv} functions is therefore in acid-sensitivity, reflected in the fitted value for CL_{Smax}. For example, the F4.2 dry heath in the Peak District Dales SAC appears to be more acid-sensitive than the F4.2 dry heath in the North Pennine Moors SAC (Figure 6.4). The reasons for these differences in acid-sensitivity are discussed in section below.

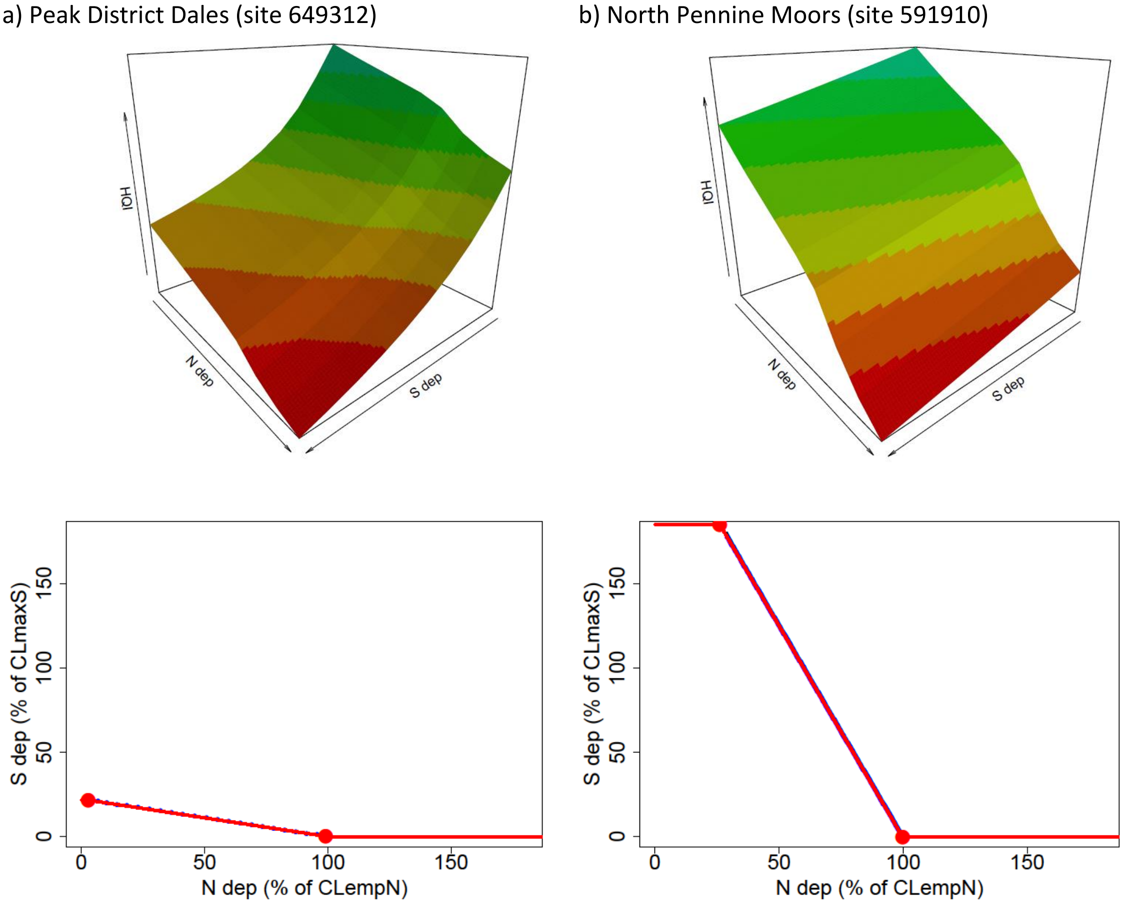


Figure 6.4. Two dry heath (EUNIS F4.2) sites with differing sensitivity to N and S pollution: a) Peak District Dales SAC; b) North Pennine Moors SAC. The upper plots show response surfaces, calculated using the MADOC-MultiMOVE-HQI model chain, for how habitat quality is affected by variation in N or S deposition. The lower plots show Critical Load functions derived from these response surfaces by assuming that the *HQI* value reaches a critical threshold at [zero non-marine S deposition, 100% of the empirical CL for N].

The MADOC-MultiMOVE-HQI model chain successfully calculated CL_{biodiv} functions for 103 Bog, 39 Dry acid grassland, 107 Wet heath and 106 Dry heath sites. These sites are mapped according to the value for CL_{Smax}, i.e. the sulphur deposition at the upper of the two points of the biodiversity-based critical load function, firstly in terms of percentage of the NFC value for CL_{maxS} (Figure 6.5), and secondly in terms of absolute value (Figure 6.6).

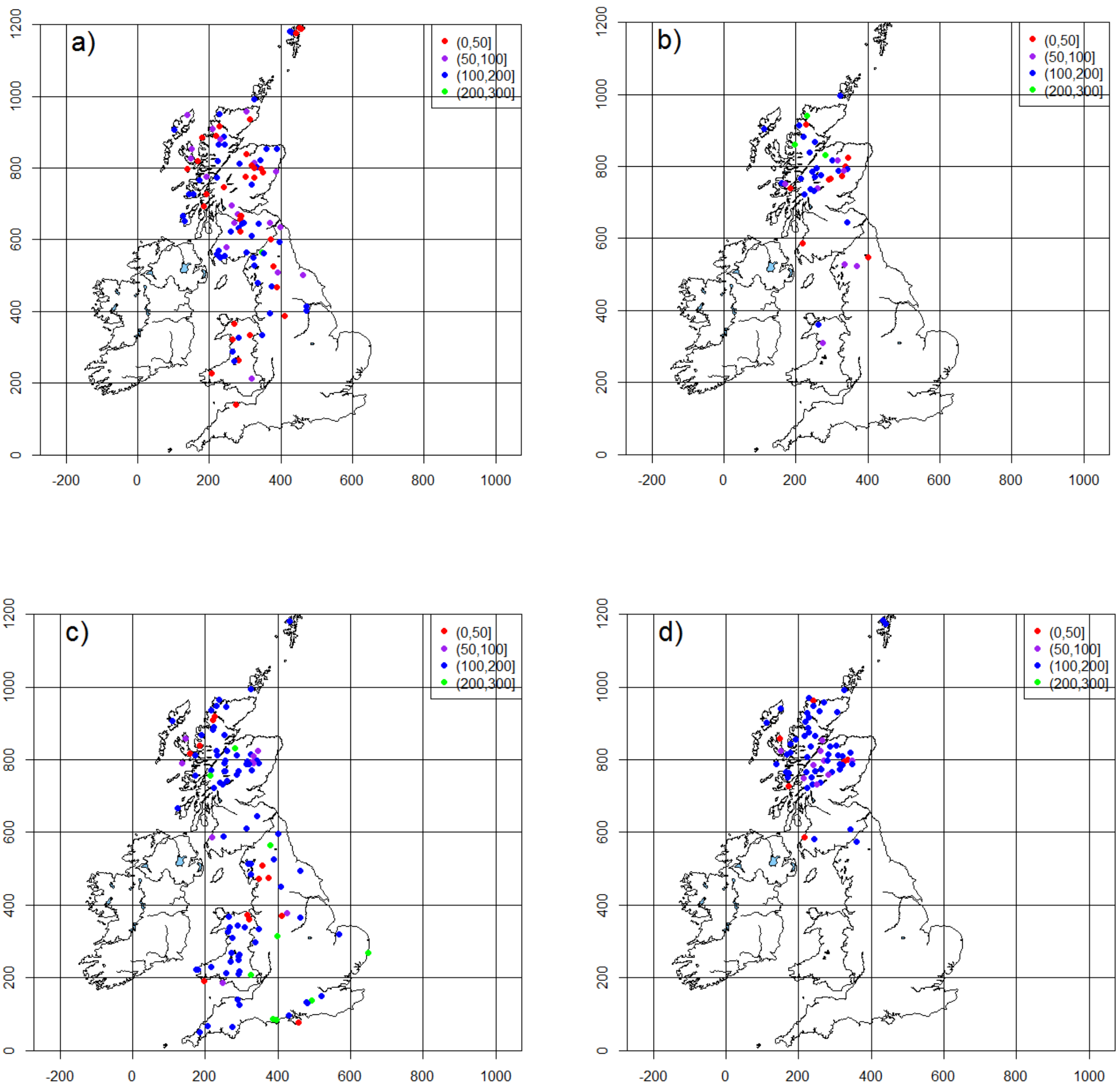


Figure 6.5. CL_{maxS} values as percentage of empirical CL_{Smax} values. Special Areas of Conservation (SAC) with a) bog (EUNIS D1); b) dry acid grassland (EUNIS E1.7); c) dry heath (EUNIS F4.2); and d) wet heath (EUNIS F4.11), for which biodiversity-based Critical Load functions have been calculated. Colours indicate CL_{maxS}, expressed as a percentage of CL_{Smax}, i.e. the empirically-based value for maximum sulphur deposition compatible with protecting the habitat at the site.

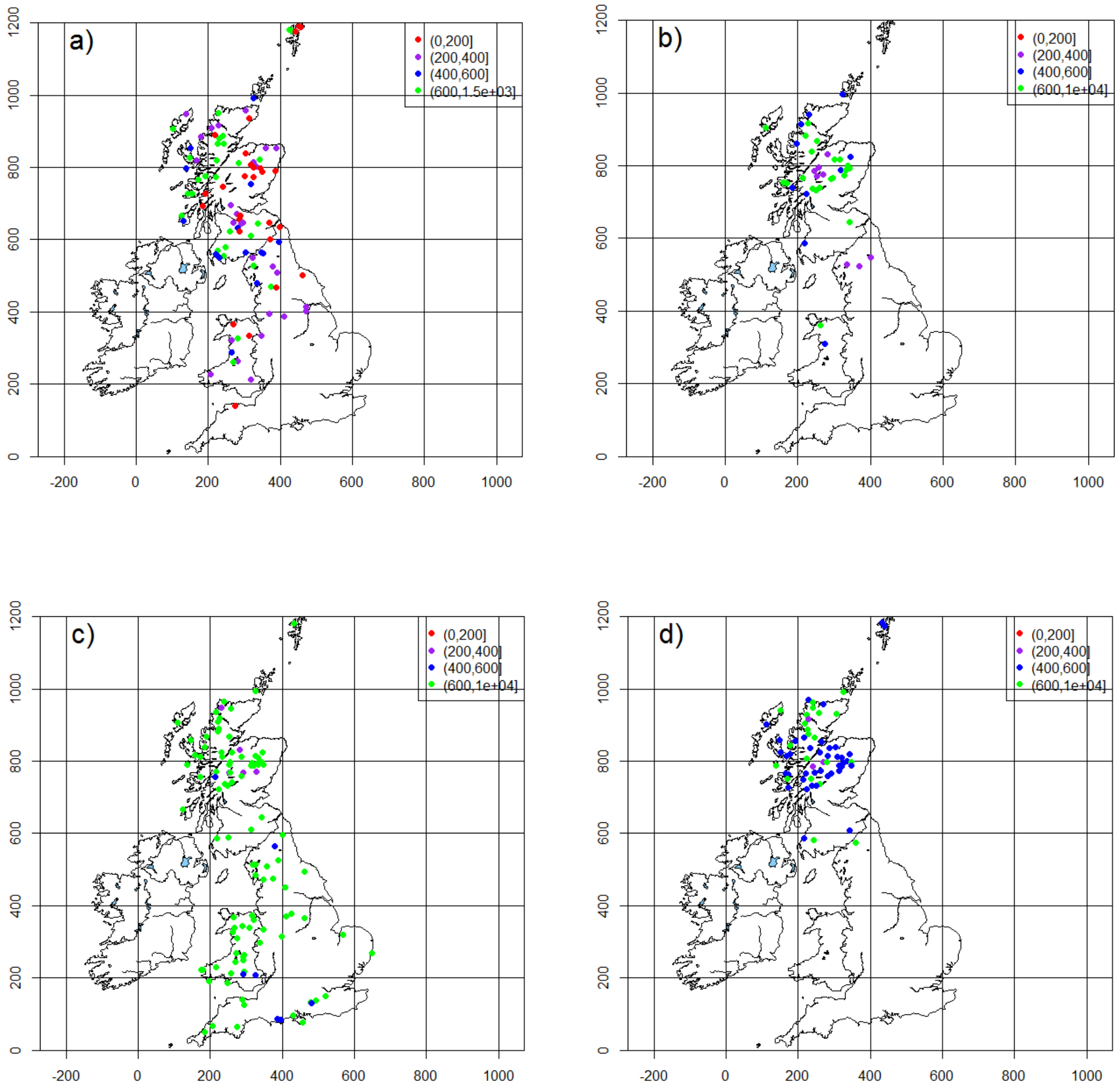


Figure 6.6. Absolute CL_{maxS} values. Special Areas of Conservation (SAC) with a) bog (EUNIS D1); b) dry acid grassland (EUNIS E1.7); c) dry heath (EUNIS F4.2); and d) wet heath (EUNIS F4.11), for which biodiversity-based Critical Load functions have been calculated. Colours indicate CL_{maxS}, expressed as eq ha^{-1} yr^{-1} i.e. as absolute values. Sites with low CL_{maxS} (red or purple) will be exceeded at low rates of S deposition compared to sites with high CL_{maxS} (blue or green).

Causes of variation in biodiversity-based critical load functions

Much of the variation observed (Figure 6.5) in the biodiversity-based CL_{Smax} when expressed as a percentage of the empirically-based CL_{maxS} values is due to the variation in the latter. The empirically-based CL_{maxS} values are derived from a simple mass balance:

$$CL_{maxS} = CLA + BC_{dep} - BC_u$$

Where:

- CLA = acidity critical load, for non-woodland habitats this is mainly based on the weathering rate for the dominant soil in each 1km square (a different calculation is used for peat), for woodland habitats a mass balance equation is used but soil parameters are based on the dominant soil.

- BC_{dep} = non-marine base cation deposition minus non-marine chloride deposition to 'moorland' (i.e. low growing vegetation) or woodland.
- BC_u = base cation uptake or removal, which is set to zero for acid grassland, dwarf shrub heath, bog & montane. (Values are assigned for calcareous grass and managed/productive woodland).

The weathering rate for different soil types varies considerably (from 100 to 4000 eq ha^{-1} year^{-1}), and variation in the empirically-based CL_{maxS} values results from the selection of different soil types as the dominant type within the 1km square. This is illustrated in Figure 6.7a) for the sites for which CL_{biodiv} functions have been generated. Some dry acid grassland and heathland sites have empirically-based CL_{maxS} values of around 4000 eq ha^{-1} yr^{-1}, reflecting a calcareous soil type. In fact the acid-sensitive habitat is unlikely to occur on this soil type. To improve the national data it would be necessary to create more detailed maps by overlaying the national soils data, for example as vector polygons, over detailed land cover polygons to create more realistic soil/habitat combinations and then apply appropriate methods to calculate CLs for each combination.

a) **b)**

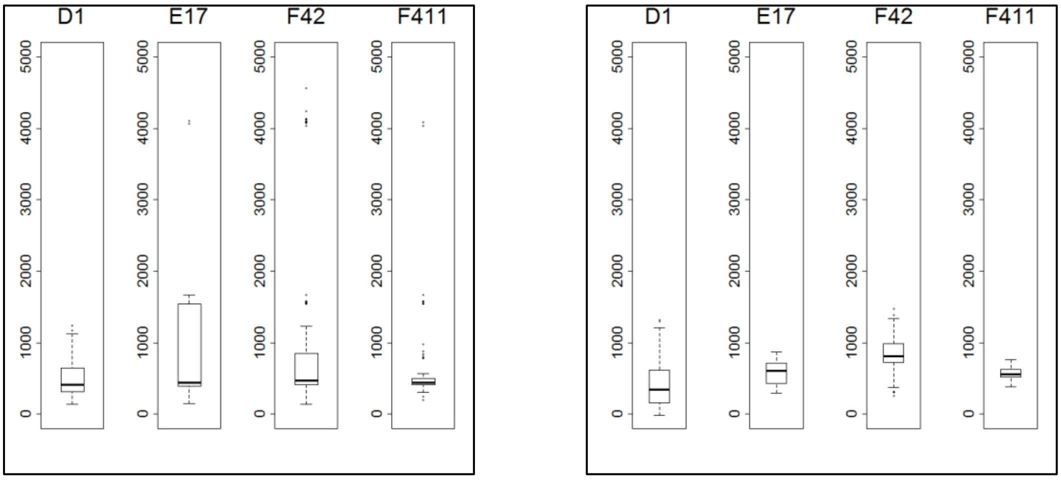


Figure 6.7: Variation in: a) the empirically-based CL_{maxS} values that are derived using dominant soil type by the National Focal Centre; and b) biodiversity-based CL_{Smax} values derived using the MADOC-MultiMOVE model chain, for four acid-sensitive habitats: bog (D1); dry acid grassland (E17); dry heath (F42); and wet heath (F411).

The variation in biodiversity-based CL_{Smax} values illustrated in Figure 6.7b) is presumably more realistic, and reflects differences in the species selected as locally-occurring positive indicator-species, and/or to differences in responses to acid deposition load of the environmental factors used to define niches in MultiMOVE, such as pH, available-N flux or canopy height. Variation in the bog and heath examples is smaller, but variation in dry acid grassland is greater, than variation in empirically-based CL_{maxS} values for the same habitats. In the following section we explore the causes of differences in acid-sensitivity between two dry acid grassland examples: the acid-sensitive Carn nan Tri-Tighearnan site (NFC code 325084) for which the biodiversity-based CL_{Smax} value was calculated to be 310 eq ha^{-1} yr^{-1}, and the relatively acid-insensitive Beinn Iadain and Beinn na h`Uamha site (NFC code 380971) for which the biodiversity-based CL_{Smax} value was calculated to be 867 eq ha^{-1} yr^{-1}.

There were only minor differences in the lists of species selected as locally-occurring positive indicators for these sites (Table 6.4). Two species (*Persicaria vivipara* and *Viola lutea*) that occurred in the 10 x 10 km hectad for the acid-sensitive Carn nan Tri-Tighearnan site did not occur in the 10 x 10 km hectad for the relatively acid-insensitive Beinn Iadain and Beinn na h`Uamha site, and one species (*Coeloglossum viride*) occurred near the relatively acid-insensitive but not the acid-sensitive site, and 30 species were common to both sites. None of these three species was strongly affected by increased acidity. Although the inclusion or exclusion of particularly sensitive species could affect acid-sensitivity in some cases, this did not cause the difference in sensitivity between these two sites. Habitat-suitability was not affected strongly by increased acidity for many of the species, and the response of mean habitat-suitability was mainly driven by relatively strong declines in suitability for a few species: *Angelica sylvestris*, *Galium verum*, *Campanula rotundifolia* and *Thymus polytrichus*.

Table 6.4: Positive indicator-species for dry heath that occur at or near a relatively acid-sensitive site (Carn nan Tri-Tighearnan) and a relatively acid-insensitive site (Beinn Iadain and Beinn na h`Uamha). Absolute changes in projected habitat-suitability in 2100 are shown for each species between scenarios with zero S deposition and S deposition at approximately 300 eq ha^{-1} yr^{-1} (270 eq ha^{-1} yr^{-1} at Carn nan Tri-Tighearnan and 318 eq ha^{-1} yr^{-1} at Beinn Iadain and Beinn na h`Uamha).

Positive indicator species	Change at sensitive site	Change at insensitive site
Anemone nemorosa	-0.001	-0.026
Angelica sylvestris	-0.115	-0.154
Calluna vulgaris	-0.031	-0.041
Campanula rotundifolia	-0.077	-0.095
Carex caryophyllea	-0.001	-0.003
Carex panicea	0.014	0.016
Cerastium fontanum	-0.011	-0.013
Coeloglossum viride	Does not occur	-0.018
Danthonia decumbens	0.000	0.000
Erica cinerea	-0.002	0.003
Erica tetralix	0.005	0.006
Filipendula ulmaria	-0.004	-0.033
Galium saxatile	-0.001	0.000
Galium verum	-0.067	-0.108
Geum rivale	-0.002	0.000
Gymnadenia conopsea	-0.005	-0.004
Lathyrus linifolius	-0.007	-0.013
Lotus corniculatus	-0.002	-0.001
Pedicularis sylvatica	0.000	-0.004
Persicaria vivipara	0.000	Does not occur
Pilosella officinarum	0.000	0.000
Pinguicula vulgaris	-0.001	-0.003
Polygala serpyllifolia	-0.028	-0.027

Polygala vulgaris	-0.007	-0.025
Potentilla erecta	-0.010	-0.014
Rumex acetosella	0.000	0.000
Succisa pratensis	-0.001	-0.001
Thymus polytrichus	-0.069	-0.064
Vaccinium myrtillus	0.001	0.004
Veronica officinalis	-0.044	-0.060
Viola lutea	0.000	Does not occur
Viola palustris	-0.065	-0.052
Viola riviniana	-0.013	-0.017

Transfer functions

Single-predictor equations

Ellenberg R was related to pH in a non-linear fashion whereby Ellenberg R was more strongly related to pH at low pH (Figure 6.8a). Above pH ~5.5 there was little evidence of a relationship between pH and community mean Ellenberg R. The data was best explained by a breakpoint (cf. broken stick) model with a breakpoint at pH 5.39. Although the conditional R^2 for this model was high (0.83) the model had relatively little predictive power when the random effect structure was not considered (R^2 = 0.17). AIC comparison showed that Ellenberg F was best predicted by VWC, and models using VWC as predictors also showed fewer patterns in residuals. The fit of the data to the model was higher than for Ellenberg R, with a marginal R^2 of nearly 0.50 (Table 6.5), however only the quadratic term in the model was significant (VWC coefficient 0.369, s.e. 0.312; VWC^2 coefficient 1.912, s.e. 0.283) suggesting that the relationship between Ellenberg F and VWC is not particularly strong, this is evident in Figure 1b where Ellenberg F is more or less constant up to VWC of ~0.6. Of the five potential predictors of Ellenberg N, organic carbon content (LOI) was the best predictor, although similarly to Ellenberg R, the R^2 was low when random effects were not considered (Table 6.5). Both linear and quadratic terms were significant (LOI coefficient -0.043, s.e. 0.002; LOI^2 coefficient 0.0002, s.e. 0.00002).

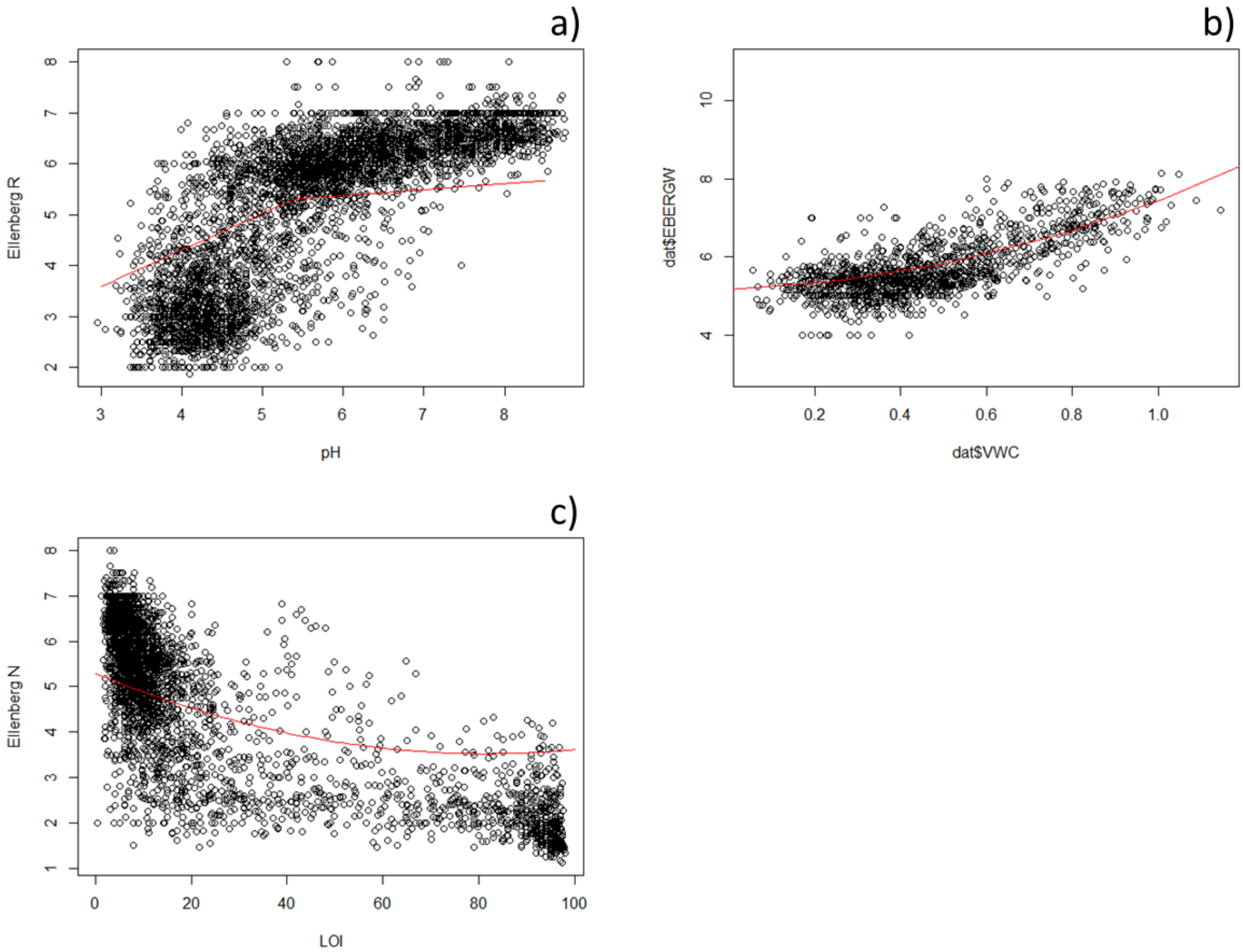


Figure 6.8: Model fits from single predictor equations for a) Ellenberg R, b) Ellenberg F, c) Ellenberg N.

Table 6.5: Coefficients and predictive power of single term models

Ellenberg indicator	Intercept	Coefficients	Marginal R^2	Conditional R^2
R	5.301	-0.717*pH < 5.39 + 0.118*pH > 5.39	0.168	0.825
F	0.142	$0.369*VWC + 1.912*VWC^2$	0.489	0.703
N	1.103	$-0.043*LOI + 0.0003*LOI^2$	0.169	0.801

Multiple-predictor equations

Selection of model terms by AIC preferred the inclusion of all 10 possible terms (five main effects plus quadratics) in both the Ellenberg R and N models, with only two quadratic terms removed for the Ellenberg F model (Table 6.6). Inspection of model summaries confirmed that almost all terms had significant coefficients in the final model. Model fit by marginal and conditional R^2 was much better than the single term models, particularly for marginal R^2 values and a large part of variance was explained (Figure 6.9).

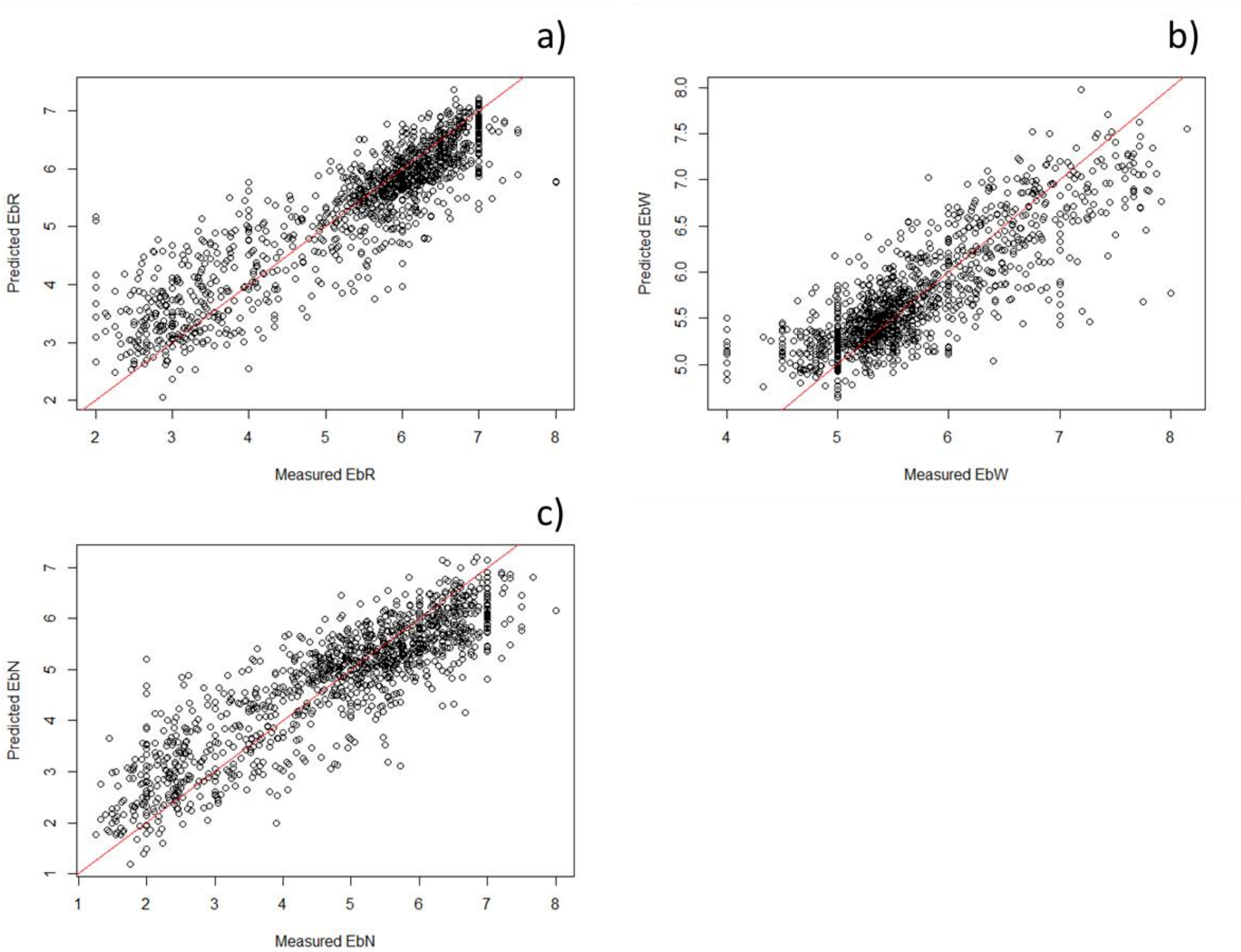


Figure 6.9: Model fits from multiple predictor equations for a) Ellenberg R, b) Ellenberg F, c) Ellenberg N

Table 6.6: Coefficients and predictive power of multiple term models

Ellenberg indicator	Intercept	Coefficients	Marginal R^2	Conditional R^2
R	1.367	1.445*pH – 0.341*N – 0.139*CN + 0.009*P + 0.310*VWC – 0.080*pH^2 + 0.048*N^2 + 0.002*CN^2 – 0.00002*P^2 – 1.266*VWC^2	0.776	0.852
F	5.309	-0.108*pH + 0.151*N + 0.043*CN - 0.003*P + 0.543*VWC – 0.0004*CN^2 + 0.00005*P^2 + 1.169*VWC^2	0.665	0.756
N	1.939	1.085*pH – 6.832*N – 1.258*CN + 1.589*P + 1.051*VWC – 0.005*pH^2 + 0.138*N^2 + 0.002*CN^2 - 0.00004*P^2 – 2.318*VWC^2	0.730	0.821

Alternative biogeochemistry-species-biodiversity models

Several other models are being applied by different signatory parties to respond to the current Call for Data. The default model being applied at European scale by the RIVM-Alterra team at the CCE is based on the PROPS species-niches model (Posch et al., 2015). This is based on a similar principle to

MultiMOVE, i.e. niches are fitted to occurrence data, but the predictor variables are abiotic measurements rather than trait-means as in MultiMOVE. There are pros and cons to this approach – it avoids the uncertainty in relating abiotic measurements to trait-means, but only data where soil measurements have been made alongside species records can be used, reducing the number of species that can be modelled. The PROPS model has used data from several European countries and so is widely applicable. Because of inconsistencies in methods for measuring plant-available N, the abiotic measurements used in PROPS to indicate N availability are total C/N and current N deposition rate. This contrasts with the UK approach, where the large Countryside Survey dataset of soil N availability has enabled construction of niche models that are responsive to N available over intermediate timescales (Rowe et al., 2012; Rowe et al., 2011a). Since PROPS does not respond directly to soil N availability, and has been constructed using data from areas in Europe with high N deposition rates, it predicts rapid declines in habitat quality with decreases in N deposition rate below 10 kg N ha^{-1} yr^{-1} – see Posch et al. (2015) Fig. 3.2, reproduced below (Figure 6.10). Many areas in the north and west of the UK receive less than this, and are generally considered to be in more favourable rather than less favourable condition with respect to N deposition. For this reason, and because the MADOC biogeochemical model incorporates processes that are important in upland soils such as pH – dissolved organic carbon interactions, we consider that the performance of the MADOC-MultiMOVE model chain compares favourably. It would however be instructive to compare the performance of different models at UK sites.

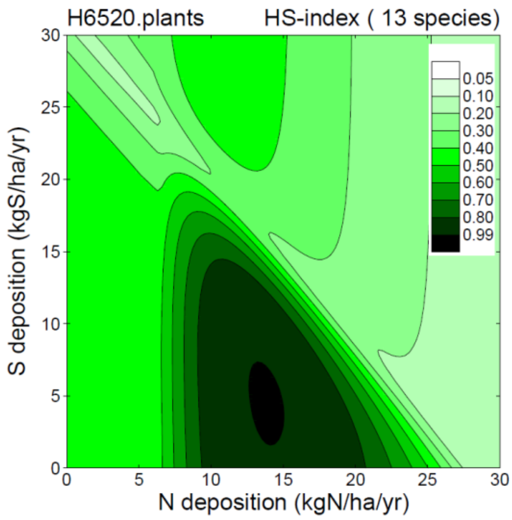


Figure 6.10: Response to N and S deposition of normalized Habitat Suitability Index for 'Mountain hay meadows' as predicted by the PROPS model (C:N = 22 g g^{-1}, T = 7 °C, P = 700 mm yr^{-1}), using the SMB model (with 'average' parameters) to link pH and S deposition. Reproduced, with permission, from Posch et al. (2015).

6.3.4 Discussion of biodiversity-based critical load functions

Performance of the model chain

The MADOC-MultiMOVE-HQI model chain has largely been automated, in that a single 'R' script is now used to run the different parts of the chain and indeed to generate and plot the biodiversity-based CL function for each site. The model chain can now potentially be applied to very many sites, and work is ongoing to allow it to be run on parallel processors so that it can feasibly be upscaled to run every 1km square in the UK with acid- or N-sensitive habitat. This capacity will be necessary to respond to the current CCE Call for Data by the deadline, which may be brought forward to January 2017. The CL_{biodiv} functions generated mainly show declines in habitat quality in scenarios where the N and/or S deposition rates increase from zero. The steepness of these declines with increasing N and S deposition rate is likely to reflect the sensitivity of the site to N and S, respectively. This sensitivity can be represented by two points corresponding to a simplified function (Figure 6.2). The model chain therefore performed satisfactorily, although there is considerable potential for improvement as discussed below.

The approach to calculating a critical threshold for habitat quality, using a scenario where N deposition was held at the CL_{empN} value from 1980 to 2100 and S deposition was held at zero non-marine deposition for the same period, was a pragmatic solution to the question of how this threshold should be established. An alternative method might be to consult with habitat specialists from the Statutory Nature Conservation Bodies as to what *HQI* value, or what corresponding artificial species-set – *cf*. Table 20 in Rowe et al. (2014a) – reflected the point at which damage occurred. In practice this would be problematic, due to the difficulty of explaining the task and providing sufficient information to describe the results of different scenarios without overwhelming the participants. The CL_{empN} values were agreed after extensive discussion and revision (Achermann and Bobbink, 2003; Bobbink and Hettelingh, 2011; Hall et al., 2011) and it is reasonably certain that they correspond to a deposition rate at the threshold of that which causes damage either immediately or in the long-term. This implies that the approach to setting the threshold is acceptable, but uncertainty remains as to a) what level of S deposition should be used for this scenario, and b) the period for which the scenario should be run. The rate of zero non-marine S deposition was chosen to avoid including effects of S on *HQI*, but in fact the values for CL_{empN} were established in experiments and surveys that were mainly carried out during the period of high S deposition rates, so it could be argued that including some anthropogenic S deposition would be more realistic. The period of 1980-2100 was chosen principally because the CCE set the date of 2100 as the end date for assessing scenarios for the Call for Data responses. The N simulated by MADOC as accumulating in soil persists for many years, which is likely to reflect the persistence of effects of N pollution in real ecosystems, as discussed in Rowe et al. (2014c). Since CL values are intended to reflect habitat protection in the long term, a start date of 1980 was chosen to extend the period of constant deposition before 2100, and to avoid including some of the effects of the late 20th century peak in N deposition.

The CL_{biodiv} functions were generated by running 100 scenarios in which N and S deposition were varied in the range 0-180 % of the empirical CL for N (CL_{empN}) and 0-180 % of the empirically-based maximum CL for S (CL_{maxS}) values for the site, as collated by the UK National Focal Centre. The CL_{empN} varies comparatively little (or not at all) among sites with the same habitat, and as noted above can be considered fairly reliable. However, the CL_{maxS} values are based on an assumption that the habitat occurs on the dominant soil type within the 1km grid square, and are strongly influenced by the base-cation weathering rate assumed for this soil type. The CL_{maxS} values can therefore vary by around ten-fold (Figure 6.7a), whereas in reality acid-sensitive habitats are unlikely to occur on soils with large base-cation weathering rates. This is a long-standing issue that has been highlighted by the NFC, but resolving it would require considerable work to overlay the UK soil and habitat maps and re-create national critical loads maps and data sets for all habitats.

Because of the large variation in CL_{maxS}, the approach to assessing the variation in *HQI* response by applying scenarios of 0-180 % of CL_{maxS} was found to be somewhat unsatisfactory. The actual value for S deposition at 180% of CL_{maxS} for the ‘sensitive’ site described in Table 6.4 was similar to the actual value for S deposition at 20% of CL_{maxS} for the ‘insensitive’ site, making it difficult to distinguish the sites’ true sensitivities from the effect of the very different CL_{maxS} values. It would be more informative to vary the S deposition in these scenarios over a consistent range for all sites, based for example on the current observed range in present-day S deposition.

Transfer functions

Our work so far in understanding the relationship between Ellenberg scores and corresponding abiotic variables indicates that this relationship is a significant cause of uncertainty in the model chain. However, the alternative is to build species niche models directly with respect to biogeochemical data, which means that only datasets where soil measurements have been made at the same time and in the same place as species records can be used. This is the approach used for the PROPS model (Posch et al., 2015). Since soil N availability is usually either not measured or measured using disparate methods, the PROPS developers are using a combination of soil total C/N ratio and current N deposition as the indicators of N saturation. This means that the model is driven largely by N deposition rate and thus responds instantaneously to changes in N deposition, taking little account of buffering.

The best fitting transfer functions seem to be those where multiple abiotic predictors are allowed to predict the Ellenberg scores. In future work, we aim to refine the predictive models and quantify the sensitivity of the model chain to this step. Key questions are:

- What is the inference from the inclusion of many terms in the multiple predictor models? Terms were selected using AIC, so in theory should be robust to the inclusion of unnecessary terms, and almost all coefficients were significant in the model summary.
- Is it necessary to include habitat-type as a predictor, for example by fitting the transfer functions separately using subsets of the data? The strong overall correlation between mean Ellenberg R and N scores makes it particularly difficult to account for low-fertility calcareous sites, although the addition of data from calcareous grassland sites has moderated this (Figure 6.11).
- What is the best approach for “reversing” the models? Most of the relationships selected using AIC included quadratic terms, and interactions between predictor variables, which are difficult to use in inverse mode. A better approach might be to fit models separately, using mean Ellenberg scores as predictors for soil variables.
- Would a more complex model fitting method (e.g. Bayesian) provide more information on uncertainty?

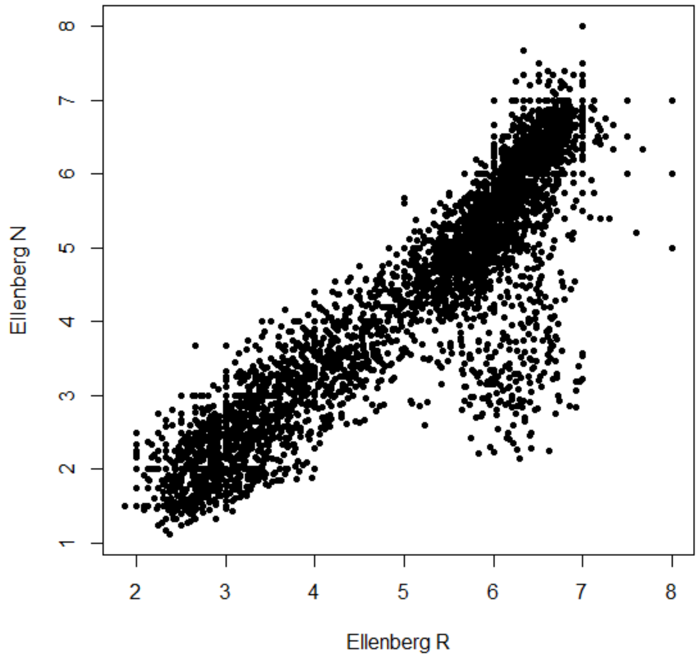


Figure 6.11: Relationship between mean Ellenberg 'R' (alkalinity) and mean Ellenberg 'N' (related to productivity or eutrophication) in the training dataset. The points with relatively high mean Ellenberg 'R' and relatively low mean Ellenberg 'N' correspond to calcareous grassland sites.

6.3.5 Conclusions

The MADOC-MultiMOVE-HQI model chain has successfully been automated, and can now be applied much more easily to generate CL_{biodiv} functions for many sites. Many uncertainties remain in the model chain, and work is ongoing to resolve and reduce these uncertainties as far as is possible. It was not possible to generate CL_{biodiv} functions for 10-20 % of sites due to missing values in the underlying datasets, but substitution of data from nearby or otherwise similar sites is likely to allow many of these gaps to be filled in future. The overall form of most of the CL_{biodiv} functions was satisfactory, showing declines in overall habitat quality as reflected by *HQI* with both N and S deposition. More confidence in the approach and in the values calculated for the CL_{biodiv} functions is likely to be obtained by running the underlying sensitivity scenarios using a more realistic range in S deposition.

7. Work Package 7: Project Reports

Summary

The key reports produced under this contract are:

- *A "Methods Report" on the calculation and mapping of critical loads and their exceedances in the UK, including site relevant critical loads.*
- *A "Trends Report" (updated in May 2016) providing a summary of the calculation of exceedances and presenting trends in (i) critical loads for UK habitats sensitive to acidification and/or eutrophication; (ii) exceedances of SRCL for UK SACs, SPAs and SSSIs.*

Both reports are available to download from the project website: http://www.cldm.ceh.ac.uk/

The following reports have been produced and submitted to Defra:

- A "Methods Report" (Hall et al, 2015a) on the calculation and mapping of critical loads and their exceedances, including site relevant critical loads.
- A separate "Trends Report" (Hall et al, 2015b and updated version in May 2016) on the calculation of exceedances and presenting the trends in critical load exceedances for broad habitats. This report now needs to be updated with the updated exceedance statistics described in Section 3.
- An Interim Report (Hall et al, 2014b; submitted February 2014) for the first project year.
- An Interim Report (Hall et al, 2015e, submitted May 2015) for the second project year.

The "Methods Report" and the "Trends Report" are available to download from the project website: http://www.cldm.ceh.ac.uk/publications/uk-status-reports#overlay-context=uk

Other activities have been reported separately:

- Analysis of deposition levels required to reduce N impacts on conservation sites (Hall, 2013), submitted to Defra October 2013.
- Report on simulation of concentration and deposition of pollutants with future shipping scenarios and assessment of ecosystem and human health impacts (Dore et al, 2014), submitted to Defra via AMEC June 2014.
- Deposition Model Inter-comparison Exercise: CBED vs FRAME (Hall et al 2015c), submitted to Defra July 2015.
- Review (Mills & Hall, submitted to Defra April 2016) of AECOM report "Developing pollutant absorption estimates for UK Natural Capital estimates: Interim Results".

8. Work Package 9: Deposition scenarios

Summary

- *FRAME deposition data sets were generated for 2025 and 2030 based on the UEP45 emissions scenario. Critical load exceedances for UK habitats sensitive to acidification and/or eutrophication were calculated for both scenarios, and both gave very similar results with less than 1% difference in the UK habitat area exceeded, and virtually the same AAE. The 2025 scenario showed a 6% reduction in the UK area of acid-sensitive habitats with exceedance of acidity critical loads, and a 19% reduction in the AAE, compared with the present day (CBED 2010-12 data, prior to 2015 update). For nutrient nitrogen the reductions were very small: 0.9% reduction in the UK area of nitrogen-sensitive habitats exceeded and a 4% reduction in the AAE.*
- *FRAME deposition scenarios were generated based on 6% and 10% reductions in sulphur deposition from shipping and compared with a baseline scenario. Differences in the critical load exceedance results for all three scenarios were small; the 10% reduction in sulphur deposition reduced the UK area of acid-sensitive habitats exceeded by 0.6% (460 km^2) compared to the baseline.*

This WP reports the critical load exceedance results based on FRAME modelled deposition for 2025 and 2030. In addition, three FRAME scenarios were analysed to look at critical load exceedances in response to reductions in SO_2 emissions from shipping; this work was done in collaboration with AMEC and reported separately to Defra in June 2014. A copy of the report on the shipping scenarios is included in Appendix 2.

An automated system has been developed in FORTRAN to calculate spatially distributed future emission files (distributed and point sources) in a format ready for input to the FRAME model. The FRAME model has been used to generate deposition scenarios for 2025 and 2030 based on the UEP45 emissions scenario. This 2025 scenario predicted large national decreases (54%) in SOx deposition and NOx deposition (29%) between 2010 and 2025. NHx emissions were predicted to increase in specific areas (non-agricultural sources) resulting in a 7% increase in NHx deposition. The 2030 scenario was very similar to 2025 with further small decreases in SOx and NOx, and an increase in NHx deposition. The FRAME deposition scenarios for 2025 and 2030 were calibrated to the CBED 2009-11 deposition data. Because the deposition scenarios for 2025 and 2030 were similar, the exceedance results were also very similar: less than 1% difference in the UK area of sensitive habitats exceeded and virtually the same average accumulated exceedance (AAE) for both scenarios. Both scenarios resulted in 40% of the UK acid-sensitive habitat areas being exceeded for acidity, and 64% of the UK nitrogen-sensitive habitat areas for nutrient nitrogen. The combined AAE for all habitats was 0.26 keq ha^{-1} year^{-1} for acidity, and 0.55 keq ha^{-1} year^{-1} (equivalent to 7.7 kg N ha^{-1} year^{-1}) for nutrient nitrogen. The maps of AAE for all terrestrial habitats combined (Figure 8.1) showed lower acidity exceedances across parts of England and Scotland compared to the results for 2010-12 (based on uncorrected CBED data; see Section 2.1), while the map for nutrient nitrogen (Figure 8.1b) looked very similar to the map for 2010-12 (again, based on uncorrected CBED data for 2010-12).

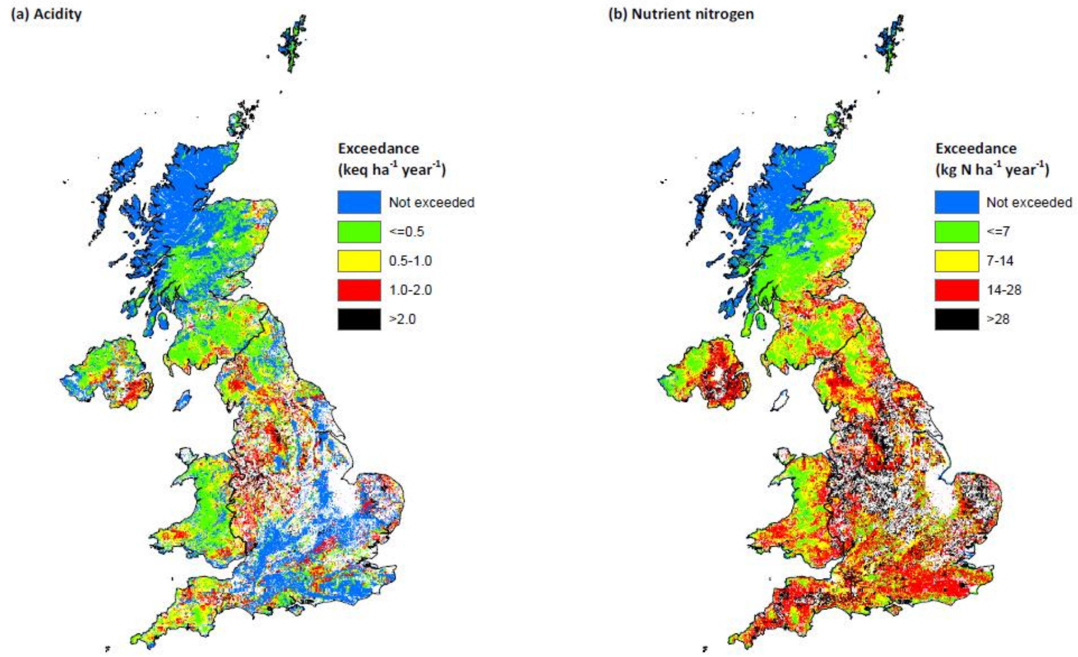


Figure 8.1: Average Accumulated Exceedance (AAE) of critical loads by FRAME deposition for 2025 (UEP45): (a) Acidity; (b) Nutrient nitrogen. Although the legends are presented in different units for acidity and for nutrient nitrogen, the class intervals are the same (eg, 7 kg N ha^{-1} $year^{-1}$ is the same as 0.5 keq ha^{-1} $year^{-1}$). The maps represent the exceedances for all habitat types mapped nationally as sensitive to acidification and/or eutrophication (WP1); areas of the UK containing other habitats to which critical loads have not been applied are shown in white.

Table 8.1 compares the 2010-12 exceedances (based on the uncorrected CBED data) and the 2025 exceedances by country. The results showed a small decline in the UK acid-sensitive habitat area exceeded and the AAE for acidity, but virtually no change in the UK exceeded area of nitrogen-sensitive habitats and a small increase in the nutrient nitrogen AAE for UK. It should be noted that this comparison was made prior to the update to CBED data to correct for the over-estimate of nitric acid deposition (Section 2.1).

Table 8.1: Critical load exceedances for CBED 2010-12 (uncorrected) and FRAME 2025 (UEP45) deposition; comparison of (a) the percentage area of habitats sensitive to acidification and/or eutrophication with exceedance of critical loads, and (b) the average accumulated exceedance (AAE).

Country	Results for acidity				Results for nutrient nitrogen			
	% area exceeded		AAE keq ha^{-1} $year^{-1}$		% area exceeded		AAE kg N ha^{-1} $year^{-1}$	
	2010-12	2025	2010-12	2025	2010-12	2025	2010-12	2025
England	65.0	65.1	0.66	0.59	97.3	97.8	16.0	17.8
Wales	76.7	70.0	0.51	0.37	91.6	91.4	10.2	10.1
Scotland	33.5	24.6	0.15	0.09	44.9	43.1	2.8	2.5
NI	68.4	67.3	0.47	0.45	89.3	90.4	10.1	10.9
UK	46.9	40.8	0.32	0.26	65.4	64.5	7.4	7.7

A comparison of the AAE calculated for individual habitats for the UK for 2010-12 and 2025, showed a decrease in the acidity AAE for most habitats (Figure 8.2), and a small increase in the acidity AAE for managed broadleaved woodland and unmanaged (other) woodland. For nutrient nitrogen there was either little change between 2010-12 and 2025, or an increase in AAE in 2025 (Figure 8.3), reflecting the increase in NHx deposition from 2010.

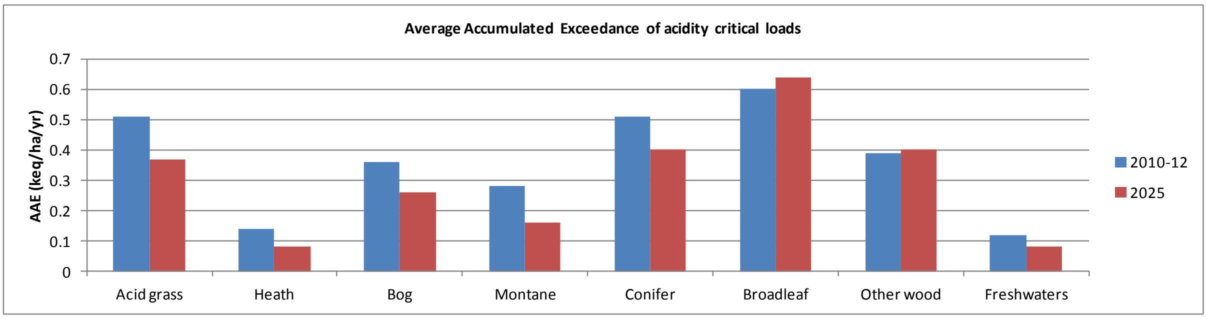


Figure 8.2: AAE for acidity by habitat for the UK, comparing results based on CBED 2010-12 and FRAME 2025 (UEP45) deposition.

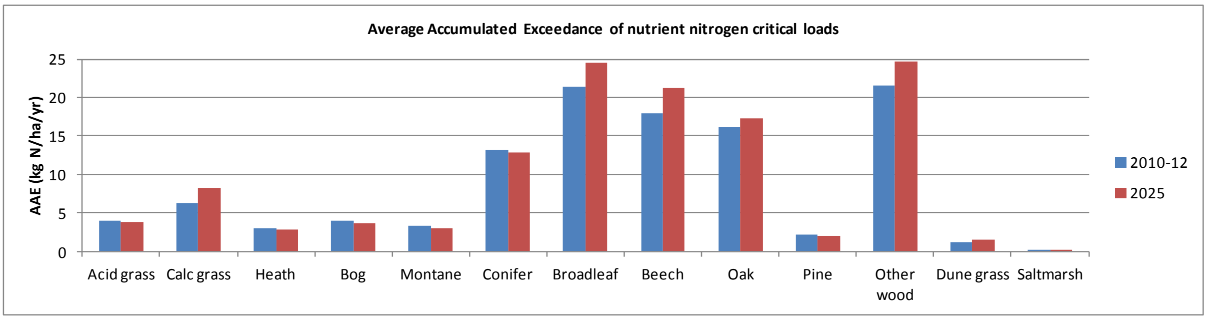


Figure 8.3: AAE for nutrient nitrogen by habitat for the UK, comparing results based on CBED 2010-12 and FRAME 2025 (UEP45) deposition.

9. Work Package 10: Development of a new method for calculating acidity critical loads for peat soils

Summary

- *A new method has been developed for calculating acidity critical loads for peat soils, based on the buffering of acidic inputs that may be provided through the reduction and subsequent incorporation of sulphate into accumulating peat organic matter. The new method is currently only applicable to upland bog habitats in good condition; for these habitats it resulted in higher critical loads and smaller or no exceedances.*
- *Peatlands that have been degraded (including those in high deposition areas of the UK) by land management activities such as burning or drainage have a reduced capacity to retain pollutants, and are therefore more vulnerable to acidification. Further work is required to develop and apply the method to degraded bog/peatland habitats, and national-scale maps of bog condition would be required to apply the new method to all bog/peatland habitat types in the UK.*
- *The new method assumes there is no net effect of nitrogen deposition on the acid neutralising capacity of peatlands and therefore only considers the acidification risk from sulphur deposition. However, this does not mean that bog/peatland habitats are not at risk from the impacts of nitrogen deposition; the empirical critical loads of nutrient nitrogen should still be applied to bog habitats in the UK.*

This section describes the characteristics of peat soils and the reasons for considering a new, alternative method for calculating acidity critical loads, as well as presenting the new proposed method and preliminary results. The new method focused on the buffering of acidic inputs that may be provided through the reduction and subsequent incorporation of sulphate into accumulating peat organic matter. It assumed there was no net effect of nitrogen deposition on the acid neutralising capacity of peatlands (Section 9.3) and therefore only considered acidification from sulphur deposition. As a result, in this exercise, the method was only used to derive the maximum critical load of sulphur (CLmaxS) and exceedances were based on sulphur deposition alone. The new critical loads were higher than those based on the current method and led to smaller or no exceedances. The method is currently only applicable to upland bog habitats in good condition; peatlands that have been degraded by land-management activities such as burning or drainage have reduced capacity to retain pollutants, and are therefore more at risk of acidification (and will have lower critical loads); further model parameterisation and national-scale maps of bog condition would be required to apply the method to all bog habitats in the UK.

9.1 Background

9.1.1 Biogeochemical characteristics of peats

Peats, or histosols, are soils with deep organic horizons. In England and Wales, a peat is defined as having an organic layer greater than 40 cm, in Scotland the threshold is 50 cm. Collectively, peats cover about 10% of the UK land area (Bain et al., 2011), and hold by far the largest terrestrial carbon store. The largest deposits are located in upland regions of the Northern and Western UK with high rainfall, where they cover large areas in the form of blanket bogs. Raised bogs occur predominantly in lowland areas, along with fens. Bog peats are ombrotrophic, receiving most or all of their water and nutrients from the atmosphere. With little contact with underlying mineral soils, weathering rates are effectively zero, and bog peats are therefore naturally acid. Bogs also produce large amounts of organic acids (i.e. dissolved organic carbon, DOC) contributing to their naturally low pH. However, due to the absence of base cation buffering, bog peats are also susceptible to acid deposition. The blanket bogs of the South Pennines were among the ecosystems most severely affected by acid deposition, with acidity effects on plants first noted as early as the 19th century in the area around Manchester (Smith, 1872), and clear evidence of vegetation changes linked to acidification by the 1950s (Pearsall, 1956). In particular, sulphur pollution of the South Pennines led to the severe decline and localised extinction of peat-forming *Sphagnum* species (Ferguson, 1978). This in turn is thought to have contributed (possibly along with intensive land-management) to an overall loss of vegetation cover, exposure of bare peat surfaces to erosion, gully formation and widespread severe (and in many areas ongoing) carbon loss and CO_2 emissions. Degraded peat areas have been slow to regain their *Sphagnum* cover as sulphur deposition has declined (e.g. Caporn et al., 2006), and restoration projects aimed at re-establishing blanket bog cover often involve drastic (and expensive) interventions involving the use of *Sphagnum* propagules, grass seed to provide a nurse crop, NPK fertiliser to increase nutrient levels and lime to raise pH.

There are strong indications that other peatland areas of the UK have suffered similar acidification damage to the Southern Pennines. The peats of the North York Moors have been shown to be strongly acidified (Evans et al., 2014; Battarbee et al., 2015), and bogs in the Brecon Beacons, close to the South Wales coalfields, show a similar pattern of large-scale *Sphagnum* loss and erosion as the Southern Pennines. Although the detrimental effects of acid deposition have been less dramatic in more remote parts of the UK, a previous survey of Scottish peats by Skiba et al. (1989) showed evidence of pH reductions and depletion of base saturation across large areas of Central and Southern Scotland. Current monitoring is taking place at a number of raised and blanket bog sites in Yorkshire, Lancashire and South Wales to establish the extent of impacts on these near-source regions (Monteith et al., 2014). From the available evidence, and given the low buffering capacity of bogs, it is probable that large areas of the UK's extensive peat area have been affected by acidification, and may (due to their very low buffering capacity) remain at risk from ongoing (albeit greatly reduced) levels of sulphur deposition. Because bogs are naturally highly nutrient-poor they are generally strong sinks for atmospheric nitrogen, although in some of the most polluted areas such as the South Pennines there is evidence that blanket bogs have become nitrogen saturated, contributing to acidification and eutrophication (Helliwell et al., 2007).

In contrast to bogs, fen peats receive much of their water input laterally, either from groundwater or river water. As a result they typically have naturally higher inputs of base cations, and a higher pH. Whilst there is some evidence that atmospheric nitrogen deposition can contribute to eutrophication of fen peats, in general fens are more affected by local nutrient sources such as agricultural runoff from surrounding fields. There is little evidence to suggest that any fens have been detrimentally affected by acidification. The following methodology therefore applies only to the bog peats which make up the majority (over 90%) of the UK's remaining semi-natural peat area.

9.1.2 Peats and critical loads: The problem

Standard approaches to calculating acidity critical loads for terrestrial ecosystems were based on the concept of mineral buffering of acidifying inputs. In effect, the critical load was the weathering rate; where the net input of acidifying anions in deposition exceeded the long-term base cation supply from mineral weathering, the critical load was exceeded and the soil became acidified. The essence of this buffering can be expressed through the charge balance definition of acid neutralising capacity, ANC, as:

$$ANC = \Sigma \text{Base Cations} - \Sigma \text{Acid Anions} \quad (1)$$

Where all terms are expressed in molar equivalent concentrations (normally in $\mu eq\ l^{-1}$), such that a positive ANC is indicative of a non-acidified situation (i.e. where bicarbonate is present in porewater or surface water) and a negative ANC indicates an acidified situation, where hydrogen ions and inorganic aluminium are present. For freshwaters, an ANC value of 0 or 20 $\mu eq\ l^{-1}$ was used as the critical damage threshold, based on evidence of damage to biota below these concentrations. For terrestrial critical loads, the simplest 'Skokloster' method (Nilsson & Grennfelt, 1988) applied an almost identical approach, whereby a flux of acidifying anions in deposition that exceeded the (estimated) flux of base cations from weathering was taken to indicate that the site was at risk of acidification (i.e. ANC < 0) under steady state conditions. For forests, although the more sophisticated Steady State Mass Balance (SMB) method was applied, and the critical threshold was based on the estimated ratio of calcium to aluminium in soil solution (Hall et al, 2015a), the model was still effectively based on the concept of base cation buffering from weathering.

For bog peats, however, the base cation weathering rate is effectively zero. This creates a problem for critical load estimation, because it implies that there is *no* buffering in the system, and therefore that *any* deposition of acidifying compounds onto a peatland may be acidifying, and thus ecologically damaging. If this logic were followed, then the acidity critical load for peats would be set to zero, and it would be impossible to achieve non-exceedance for the UK's large peatland area unless all acidifying emissions ceased. This is clearly not a realistic prospect, so alternative approaches are required.

9.1.3 Previous methods for calculating acidity critical loads for peat

Given the zero weathering rate problem, it was recognised in the UK that classical approaches to critical load estimation would not work for peatlands. Instead, following the original evidence of Skiba et al. (1989), an approach was developed based around pH. Smith et al. (1992) used laboratory experiments to determine the deposition load that would cause peat pH (measured in $CaCl_2$) to decrease by 0.2 units relative to 'pristine' levels. This threshold was subsequently adopted as the basis for setting critical loads for peats (Hornung et al., 1995). Subsequently, however Cresser (2000), one of the authors of the original study, suggested that the value of 0.2 pH units was 'rather arbitrary', and suggested an alternative approach based on 'effective rainfall pH'. This value, representing the total (wet plus dry) acid load divided by runoff, was suggested to be indicative of the pH of peat porewater, based on an empirical, 1:1 relationship between the two measurements recorded by Proctor and Maltby (1998).

A range of evidence relating to the ecological impacts of acidification on bog ecosystems has been used to support the critical load. Yesmin et al. (1996) observed significant declines in enchytraed worm populations along a pollution gradient, and further work by Yesmin et al. (1996) also observed a decrease in *Calluna* mycorrhizae which was correlated to effective rainfall pH. Calver et al. (2004) exposed vegetated peat mesocosms to a range of acid deposition loads and observed little evidence of negative impacts until porewater pH fell to 3.6. At this point, although *Calluna* appeared relatively resilient, negative impacts were observed on the abundance of a number of bryophyte species, including *Sphagnum*. On the basis of this and previous studies, Calver et al. (2004) recommended that an effective rainfall pH of 3.6 should be used as the critical acidity threshold for peats.

The UK National Focal Centre undertook a review of the acidity critical load method for peats in 2002 (Hall et al, 2003), and concluded that an effective rainfall pH of 4.4 should be used to set critical loads. This was based on an analysis by Cresser (2003) (reproduced in Hall, 2006) which focused mainly on acidity impacts on *Calluna*, and recommended a more precautionary threshold to take account of apparent curvature in the relationship between apparent rainfall pH and peat porewater pH, and to allow for the buffering effects of organic acids on peat drainage water pH. This threshold has remained in place until the present time.

Based on a re-evaluation of the approach taken to peat critical loads to date, we suggest that there are a number of serious, and fundamental, problems with these methods. First and foremost, peat porewater pH is an extremely difficult parameter to predict, because the pH (i.e. hydrogen ion concentration) of a solution is the outcome of a complex set of equilibrium processes involving a wide range of ions (this is one reason why acid-base models such as MAGIC and VSD focus primarily on modelling ANC rather than pH). The problem is particularly acute for peatlands, because DOC concentrations are typically high (and highly variable), and the associated organic acid concentrations have a very strong (but hard to predict) influence on porewater pH. This is discussed further below. Linked to this, there appears to be no clear mechanistic basis for the inference (from a single study) that peat porewater pH is a 1:1 function of 'effective rainfall pH'. This is partly because of the major influence of organic acidity (which is not present in rainfall) on porewater pH, but also because the pH of both rainfall and peat porewater is highly dependent on the form of nitrogen present, i.e. the balance between ammonium (NH_4^+) and nitrate (NO_3^-). Ammonium in rainfall will raise the pH, and indeed (as sulphate emissions have declined but ammonia emissions have not) the pH of rainfall in the UK has increased to the point that it is technically alkaline (RoTAP, 2012). This does not stop N deposition having an acidifying impact, if NH_4^+ is oxidised to NO_3^- and leached, but in the reducing, N-limited conditions that occur in bogs this may not necessarily be the case (see below).

Finally, the emphasis of much of the past ecological impacts work on *Calluna* is puzzling, because *Calluna* is far less important to peatland ecosystem function than other bog species, notably *Sphagnum* mosses, and almost certainly less sensitive (as was shown by Calver et al., 2004) to acidification. Indeed, *Calluna* dominance is often viewed as a negative condition attribute of bogs (e.g. Smyth et al., 2014) because it tends to increase as a result of drainage or burning. The absence of a detrimental effect of acidification on *Calluna* can therefore in no way be taken to indicate the absence of a detrimental impact of acidification on other bog species, or on ecosystem function.

9.1.4 How do peats *actually* buffer against acidification?

During the ongoing development of different critical load methods for the UK's peatlands, little attention has been given to the fundamental biogeochemical processes that occur within peatlands. This is surprising, because these processes clearly have the potential to buffer peatlands against acidification. In large part, the key processes are linked to the naturally waterlogged condition of peat soils. This waterlogging greatly limits oxygen transport into the soil, leading to anaerobic, or reducing, conditions. One consequence of these conditions is that organic matter decomposition is greatly reduced, leading to an excess of plant organic matter production over organic matter decomposition, and therefore to peat formation. This imbalance means that peats are fundamentally different to other soil types, in that they do not reach a steady state condition (where organic matter inputs and outputs are equal) for many thousands of years. During this time, they not only accumulate carbon, but also have the potential to accumulate other elements, including pollutants. Some key processes are considered below.

Sulphur retention

Sulphur is emitted from fossil fuel burning in oxidised form, and deposited on ecosystems as sulphate, SO_4^{2-}. Mineral soils can retain some sulphate via anion adsorption onto mineral surfaces, which is an important buffering process in areas such as the Southeast United States and Germany with older soils, but provides only a very minor sink for SO_4 in younger, UK soils (Hughes et al. 2012). In coastal regions with high marine ion deposition, any sulphate adsorption capacity may in any case have been saturated by marine SO_4 inputs, and in most upland catchments it has been shown that measured SO_4^{2-} outputs in runoff are approximately equal to deposition inputs on an annual basis (Cooper, 2005).

In peatlands, anion adsorption is effectively zero. However, the anaerobic conditions that prevail in waterlogged peat allow sulphate reduction, a biological process in which microbes use sulphate as an alternative to oxygen in order to break down organic matter substrates. Sulphate reduction leads to the formation of sulphides (such as iron sulphide, Fe_2S) and organic sulphur compounds. Many of these compounds remain in the peat, and over time they will be buried and effectively 'locked up' as part of the process of peat formation. Because sulphate reduction also consumes hydrogen ions, it also raises porewater pH, and effectively buffers the ecosystem against acidification. Any leaching of organic sulphur in drainage waters (as dissolved organic sulphur, DOS) will also remove sulphur from the ecosystem without an associated acidifying impact.

From a critical load perspective, only processes that can neutralise acidity under long-term 'steady state' conditions can be considered to contribute to the buffering capacity of the ecosystem. In the case of sulphate reduction, there is clear evidence from many studies that sulphur retained in peats under wet conditions can subsequently be released through oxidation under drought conditions (e.g. Adamson et al., 2001; Clark et al., 2006) or as a result of drainage (e.g. Daniels et al., 2008). The resulting pulses of SO_4^{2-}, transported into water courses, can result in severe and biologically damaging acid episodes (e.g. Aherne et al., 2006) and have been recognised as a 'legacy pollution' issue in some areas where peatlands have accumulated stores of sulphur that are prone to re-oxidation (Daniels et al., 2008; Evans et al., 2014). Despite these issues, it is clear that where peats remain wet, and where sulphur deposition remains reasonably low, peatlands effectively act as a permanent sink for a significant fraction of the total deposition load, as a result of which streams draining peat catchments consistently have lower SO_4 concentrations than nearby streams draining mineral soils (e.g. Evans et al., 2006). This long-term retention of sulphur in accumulating peats is therefore an important, but previously unquantified, sink for pollutant SO_4 deposition.

Nitrogen retention and transformations

As noted above, bogs rely on atmospheric inputs for their nutrient supply, and support plants that are highly adapted to both nutrient-poor and waterlogged conditions. Because of this, bogs are highly effective at retaining any inorganic nitrogen inputs entering the ecosystem through atmospheric deposition. As for sulphate reduction, the uptake of nitrogen into biomass effectively consumes hydrogen ions, and buffers the ecosystem against acidification. Nitrogen incorporated into plant biomass is tightly cycled within the system, and will either be retained within living biomass or accumulated in dead organic matter. As a result, nitrate leaching from peatlands is typically near-zero, implying that nitrogen is making little contribution to acidification of either the terrestrial or downstream freshwater ecosystems. Whilst some nitrogen may be exported from the system in organic form (i.e. as dissolved organic nitrogen, DON) as in the case of DOS this is not acidifying. In contrast to other terrestrial ecosystems, where much of the currently observed nitrogen retention is generally assumed to indicate the progressive enrichment of a finite pool (i.e. eutrophication, leading ultimately to nitrogen saturation, nitrate leaching and acidification), this is not necessarily the case in a peatland, because nitrogen accumulated into peat organic matter can (as for sulphur) be continuously buried through the process of peat formation. Recently collected peat core data from Wales (see Section 2) suggest that blanket bogs have accumulated nitrogen over the last 150 years at mean rates ranging from 14 to 28 kg N ha^{-1} yr^{-1} (R. Collier, unpublished data), suggesting that a healthy peat may be are able to remove most if not all of the incoming deposition N. Again, this important long-term sink for acidifying nitrogen inputs has not been explicitly considered in previous critical load methods for peatlands.

One important caveat regarding the capacity of peatlands to buffer against the acidifying impact of N deposition (and also S deposition) is that it requires that the peatland remains wet, and continues to function as a peat-accumulating system. This is critically dependent on the presence of key peat-forming species, notably *Sphagnum* mosses, which are considered to be sensitive both to acidification and to nitrogen as a nutrient. At low levels, N deposition may slightly enhance *Sphagnum* growth and peat formation (Turunen et al., 2004) but at higher deposition rates, such as those occurring across most of the UK, N enrichment can lead to increased cover of vascular plants, resulting in the progressive loss of peat forming species and therefore to reduced peat formation (Sheppard et al., 2011). This is the basis of the empirical critical load for nitrogen as a nutrient (5 to

10 kg N ha^{-1} yr^{-1}; Bobbink and Hettelingh, 2011). At high levels of N deposition this can lead to a failure of the process of N uptake and retention, partly because peat is no longer being formed. Lamers et al. (2000) found that *Sphagnum* lost the ability to retain N at around 20 kg N ha^{-1} yr^{-1}. In terms of acidification, this suggests the possibility of a 'tipping point' whereby ecological changes brought about via high N loadings lead to a failure of ecosystem function (i.e. reduced peat accumulation), a resulting loss of the capacity of the ecosystem to retain atmospheric S and N, SO_4^{2-} and NO_3^- leaching, and subsequent acidification. Thus exposure to one ecological pressure (eutrophication) may increase susceptibility to another (acidification), intensifying the overall impact of atmospheric pollutants on peatland ecosystems.

Finally, it is important to note that N is deposited on terrestrial ecosystems as a variable mixture of NH_4^+ and NO_3^-. In most ecosystems, both are rapidly incorporated into biological cycles (e.g. Curtis et al., 2011). In aerobic mineral soils, any mineral N leached from the ecosystem is likely to be in the oxidised form, as NO_3^-, which is acidifying. In waterlogged peatlands, however, redox conditions favour NH_4^+ over NO_3^- in porewaters. If this NH_4^+ is leached from the soil into watercourses it is rapidly nitrified to NO_3^-, and therefore contributes to the acidification of surface waters. Within the peat itself, however, the excess of NH_4^+ over NO_3^- that is almost invariably observed means that the net effect of mineral N in porewater is to *raise* the pH, and thus to buffer the terrestrial ecosystem against acidification. An extreme example of this was observed at the Whim experimental site, where gaseous NH_3 application has raised porewater pH from around 3.8 to 4.3, equivalent to two thirds reduction in hydrogen ion concentrations (Evans et al., 2008). Default critical load calculations and models that assume 100% nitrification of incoming NH_4^+ in peat soils may therefore be adequate for predicting acidity in drainage waters, but highly inaccurate for peat porewaters.

Organic acid buffering

As noted above, bogs produce large amounts of dissolved organic carbon (DOC), and associated organic acidity. This organic acidity exerts a strong (natural) influence on the pH of peat porewaters, representing a major problem with the use of pH to define critical limits in peatlands (indeed, this problem is more acute for peatlands than for any of the terrestrial ecosystems where pH is *not* used to set critical limits). Because DOC concentrations vary across peatlands as a function of rainfall, evaporation, temperature, vegetation and management (e.g. Monteith et al., 2015) this effect is difficult to quantify. Furthermore, there is now clear evidence of a negative feedback between rates of mineral acid deposition from the atmosphere, and the concentration of organic acids in the peat, because DOC solubility is pH dependent. This was originally claimed to lead to the complete neutralisation of acid deposition by organic soils (Krug and Frink, 1989), but has subsequently been shown to act as only a partial buffer on acidity change (Evans et al., 2012). Nevertheless, this mechanism is sufficient to significantly reduce the degree to which peat pH will change in response to a change in atmospheric S and N deposition.

Base cation deposition

In the absence of weathering, much or all of the base cation supply to bogs comes from the atmosphere. Whilst marine base cations have no net long-term effect on acidity (because they are balanced by equivalent deposition of marine chloride and sulphate), deposition of non-marine base cations, such as dust particles, may be an important source of acid neutralisation. This was recognised for example by Calver et al. (2004), and is currently included in the calculation of CLmaxS for all habitats, based on CBED estimates of non-marine calcium and magnesium deposition.

9.2 Quantifying the role of S accumulation as a buffer against acidification

During this project, an opportunity arose to utilise a set of 50 peat cores that had been collected as part of a CEH-supported PhD study by Rob Collier at the University of Southampton. Cores were collected to a depth of 50 cm from sites representing a range of vegetation cover and management impact (e.g. drainage, burning and afforestation) across four areas of Mid and North Wales, divided into thin layers, and a combination of methods using radio-isotopes and spherical carbonaceous particles (SCPs, which are formed during high-temperature coal combustion in power stations and are thus a tracer for the emissions peak of the late 20th century) used to generate a full age profile for each core. For the PhD project, cores have been analysed for carbon content and bulk density, allowing rates of peat carbon sequestration to be quantified as a function of age and peat management history. For the current project, we utilised a part of the budget allocated for the peat critical load task to analyse all of the samples collected in the project for their sulphur content. These data have been used to estimate the capacity of peats to accumulate sulphur over time, and hence to buffer against acidification by sulphur deposition.

Results from the study are shown in Table 9.1. These data showed that the Welsh blanket bogs studied have, on average, accumulated 1680 kg S ha^{-1} of sulphur in their top 50 cm. This depth usually encompasses, and sometimes exceeds, the time period which has elapsed since the onset of sulphur emissions in the early part of the industrial revolution. Taking 1850 as an indicative start date for anthropogenic sulphur emissions (this is generally considered to be the date at which SCPs first appear in the peat record), and calculating sulphur stocks above this horizon for each peat core, suggested that they have accumulated about 1100 kg S ha^{-1} during the industrial period. As noted above, however, peats in the UK also receive SO_4^{2-} from sea-salt deposition, so not all of the sulphur accumulated over this period can be considered anthropogenic in origin, or therefore indicative of buffering of acid deposition. From the core data, we estimated that Welsh blanket bogs were on average sequestering around 5 kg S ha^{-1} yr^{-1} in 1850, assumed to be mainly of marine origin. This accumulation rate then increased in response to pollutant sulphur deposition, peaking (on average across all cores) at around 16 kg S ha^{-1} yr^{-1}. This implied that the blanket bogs studied were able to retain a maximum of 11 kg S ha^{-1} yr^{-1} of anthropogenic deposition. Although the date at which this maximum retention occurred varied between cores (which may in part reflect the uncertainty involved in dating individual horizons, or could reflect movement of SO_4^{2-} within the peat column) the average date at which peak sulphur accumulation occurred across all 50 cores, 1967, coincides exactly with the peak in UK pollutant sulphur emissions.

Table 9.1: Measured sulphur stocks and accumulation rates for a set of 50 cores collected across Wales under a range of management conditions.

Peat type	Total S stock in 50 cm peat core		Total S stock accumulated since 1850		Mean S accumulation rate (0-50 cm)		S accumulation rate in 1850		Maximum S accumulation rate		Mean year of max S accumulation
	kg S ha^{-1}		kg S ha^{-1}		kg S ha^{-1} yr^{-1}		kg S ha^{-1} yr^{-1}		kg S ha^{-1} yr^{-1}		
	Mean	*SE*	Mean	*SE*	Mean	*SE*	Mean	*SE*	Mean	*SE*	
Natural mire	1504	*236*	1348	*178*	9.2	*1.2*	9.5	*1.9*	18.9	*1.6*	1950
Afforested	1782	*145*	1388	*154*	6.5	*1.2*	4.4	*1.8*	27.5	*2.7*	1986
Drained	1534	*109*	1073	*150*	6.1	*1.2*	6.4	*1.8*	16.8	*2.4*	1953
Burn-managed	2106	*174*	1267	*93*	4.7	*1.0*	3.6	*1.4*	19.4	*2.1*	1968
Eroded	1709	*180*	815	*193*	3.6	*1.0*	2.7	*0.7*	9.7	*6.4*	1983
Molinia -dominated	1697	*246*	903	*226*	4.6	*1.0*	3.7	*0.6*	11.5	*6.7*	1972
All	1682	*79*	1096	*74*	5.8	*0.5*	5.0	*0.0*	16.1	*1.7*	1967

Examining the data in Table 9.1 in more detail, there are some indications that the rate at which blanket bogs are able to sequester sulphur, and therefore to buffer against acidification, varied as a function of management. The total post-1850 stock and average accumulation rate of sulphur was highest in samples collected from natural mire sites, and markedly lower in samples collected from drained, burn-managed, eroded and *Molinia*-dominated locations. Encroachment of *Molinia caerulea*, or purple moor grass, took place in the early 20th century across many of the blanket bogs of Wales and Southwest England, and is usually associated with increases in sheep grazing, although other factors including atmospheric nitrogen deposition may also have contributed to its rise (Chambers et al., 2007). An implication of these findings is that peatlands which have been degraded by one or more management activities may be less able to form new peat, and therefore less able to accumulate sulphur or to buffer against acidification. Although we did not have data from heavily deposition-impacted peatland regions such as the Peak District or Brecon Beacons, it is likely that the capacity of highly degraded peatlands to retain sulphur will be even lower than at these Welsh sites. This suggested a potential positive feedback, whereby damaging levels of atmospheric pollution (potentially exacerbated by local land-management factors) degrade the peat-forming vegetation, reduce the capacity of the ecosystem to buffer against sulphur and nitrogen deposition, and therefore make it increasingly susceptible to further acidification and eutrophication. The implication of these findings would be that peatlands that have been detrimentally impacted by management should have a lower critical load for acidity. Similarly, there may also be a case for assigning a lower acidity critical load to peatlands where the critical load for nutrient nitrogen is exceeded to the extent that peat-formation is likely to be impaired.

9.3 Development of a new critical load method for peat

Based on the earlier assessment, we concluded that pH is unsuitable as a basis for setting critical limits for peatlands, because it is i) problematic to estimate as a function of deposition loading, and ii) strongly influenced by natural (and hard to quantify) variations in organic acidity. The selection of pH dates back to early work undertaken in the 1980s and 90s, is founded on an assumed relationship between rainfall pH and soil solution pH that does not have a strong evidential basis, and is inconsistent with the approaches used for other UK terrestrial ecosystems, with the resulting risk of inconsistencies in exceedance calculations for similar habitat types (e.g. bogs and heathlands).

As noted above, the 'Skokloster'-based (ie. empirical) critical loads used for other non-forested UK terrestrial habitats are essentially based on the base cation weathering rate, and exceedance calculations determine whether the input of acid anions in deposition exceeds this weathering rate. This balance can be expressed in terms of Acid Neutralising Capacity (ANC), also used to set freshwater critical loads, which is routinely defined in terms in terms of charge balance as:

$$ANC = Ca^{2+} + Mg^{2+} + Na^{+} + K^{+} + NH_4^{+} - SO_4^{2-} - Cl^{-} - NO_3^{-} \qquad (1)$$

Where all ions are expressed as concentrations in μeq l^{-1}, or alternatively as a flux, in meq m^{-2} yr^{-1}. By combining individual base cations (Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}) into a single base cation (BC^{n+}) term, and subtracting the neutral sea-salt component of the total deposition, this equation simplifies to:

$$ANC = xBC^{n+} + NH_4^{+} - xSO_4^{2-} - NO_3^{-} \qquad (2)$$

Where 'x' signifies the non-marine component of the total concentration, calculated by subtracting the marine component based on known ratios of ion to Cl^{-} in sea-salt. Note that this equation omits any non-marine chloride (xCl^{-}) deposition; there is evidence that this was an important cause of peat acidification in the past (Evans et al., 2011), but current xCl^{-} deposition is close to zero so it is no longer thought to contribute significantly to critical load exceedance.

In the case of peats, where weathering rates are generally considered to be negligible, the flux of xBC^{n+} can be assumed to derive entirely from non-marine base cation deposition. The remaining unknowns in equation 2 are therefore i) the flux of xSO_4^{2-} from the peat, ii) the total flux of mineral nitrogen from the peat, and iii) the fraction of this total flux that is nitrified ($f_{nitrification}$). In practice, as noted earlier, the majority of measured peat porewater data suggest that $NH_4^{+} > NO_3^{-}$, i.e. that $f_{nitrification} < 0.5$. In this case, any acidity critical loads model that incorporates inorganic N leaching will be unable to generate a useable critical loads function, because increasing the rate of N deposition (and hence N leaching) will reduce porewater acidity. A critical loads model in which increased N deposition reduced peat acidification risk would clearly be inconsistent with the desire to protect ecosystems from the impacts on N as a nutrient, and also inconsistent with the protection of downstream water bodies, which would (after the NH_4^{+} leached from the peat has nitrified to NO_3^{-}) be at high risk of acidification due to inorganic N leaching. On this basis, we proposed that the acidity critical load model for peat should only consider the acidifying impact of S deposition (this implicitly assumed no net effect of N deposition on ANC, which is reasonable in most peatlands where inorganic N leaching is negligible.). The ANC equation then simplifies further to:

$$ANC = xBC^{n+} - xSO_4^{2-} \qquad (3)$$

The rate of xSO_4^{2-} leaching (as a flux, in mmol m^{-2} yr^{-1}) can be determined as the balance between S sources and S sinks, as:

$$xSO_4^{2-} = S_{dep} - S_{acc} - DOS \qquad (4)$$

Where S_{dep} is the total (marine plus anthropogenic) S deposition, S_{acc} is the rate of S accumulation into peat, and DOS is the leaching of S in organic form, which is assumed to be unreactive and therefore not to contribute to acidification. Based on the peat core S data analysed in Section 9.2, we derived a simple response function to describe S_{acc} as a function of S deposition, of the form shown in Figure 9.1, which assumes that peat S accumulation will occur at background levels (with no SO_4^{2-} leaching) until a lower threshold (S1) is reached, at which the accumulation rate begins to increase, and some SO_4^{2-} leaching may also occur. This rate is assumed to increase linearly until an upper threshold (S2) is reached, at which point the capacity of the peat to accumulate S is saturated, and all additional S deposition will be leached.

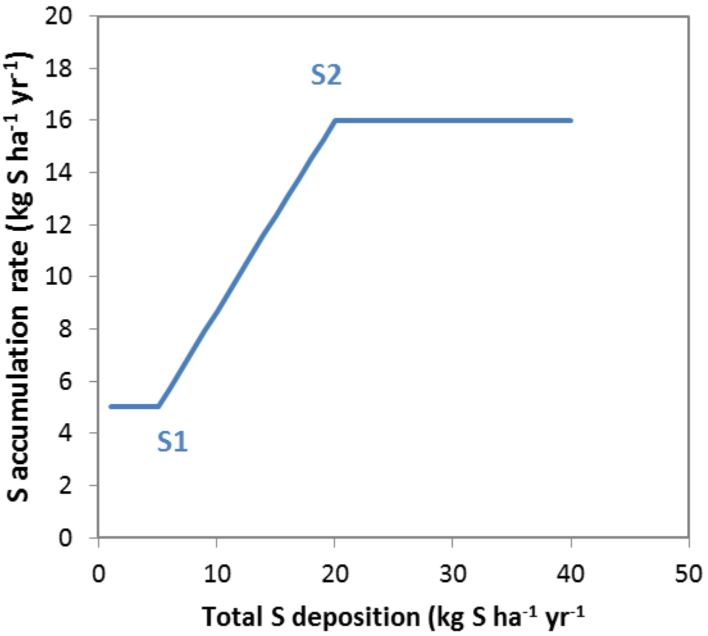


Figure 9.1: Response function for the relationship between peat S accumulation rate and total (marine plus non-marine) S deposition, based on peat core data (Section 9.2). Point S1 represents the S deposition at which the peat S accumulation begins to rise above background rates, and point S2 represents the S deposition at which the maximum rate of peat S accumulation occurs.

As noted above DOS leaching represents a small but potentially significant alternative loss pathway for deposited S, which will not contribute to acidification. The rate of DOS leaching was assumed to be a function of the rate of the DOC/DOS ratio in porewaters (DOC was assumed to be a fixed flux). The DOC/DOS ratio was in turn assumed to be the same as the C/S ratio of newly formed peat, which that core data analysed in Section 9.2 showed to vary as a function of S deposition. This was therefore described via a response function similar to that for S accumulation, shown in Figure 9.2.

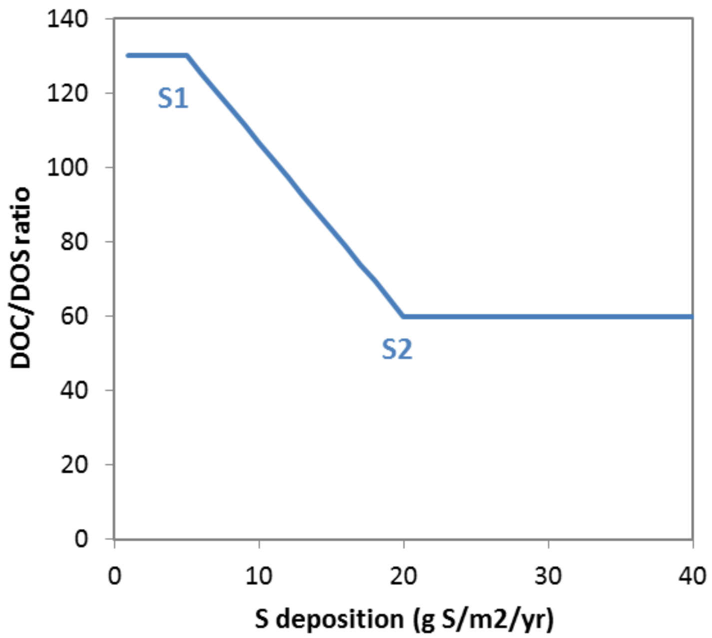


Figure 9.2: Response function for the relationship between peat porewater DOC/DOS ratio and total (marine plus non-marine) S deposition, based on peat core data (Section 9.2). Point S1 represents the S deposition at which the peat S accumulation begins to rise above background rates, and point S2 represents the S deposition at which the maximum rate of peat S accumulation occurs.

9.4 Model parameterisation and application

Most parameters for the new peat critical load model (Table 9.2) were derived from the peat core data analysed in Section 9.2, from which it was possible to determine background rates of S accumulation and peat C/S ratio, and values for each of these terms (i.e. maximum S_{acc}, minimum C/S ratio) associated with the S deposition peak of the 1960s-70s. The value of S1 was assumed to be equal to the baseline rate of S accumulation (5 kg S ha^{-1} yr^{-1}), in other words it was assumed that any increase in total S deposition above background levels would lead to an increased rate of peat S accumulation, as this was commonly observed in the peat cores. The level of total S deposition at which the peat S sink becomes saturated, S2, must (on a mass balance basis) be larger than the maximum rates of S accumulation recorded in the peat cores (16 kg S ha^{-1} yr^{-1}). On the basis that peats receiving levels of S deposition below this value generally do exhibit some SO_4^{2-} leaching – in other words that the peat S sink is never 100% effective – we provisionally set the value of S2 to 20 kg S ha^{-1} yr^{-1}. This value may need to be refined in future based on additional data. The rate of DOC leaching was considered to be fixed, and was set to a default value of 210 kg C ha^{-1} yr^{-1} which was taken from the IPCC Tier 1 default value for DOC leaching from an intact temperate bog (IPCC, 2014), which was in turn derived largely from flux studies in UK and Irish blanket bogs (Evans et al., 2016). In addition to the fixed parameters in Table 9.2, the method required the site-specific parameter values listed in Table 9.3. These were all available from the existing 1km critical loads database.

Table 9.2: Fixed parameters used in the peat acidity critical loads model

Parameter	Units	Description	Default value
$S_{acc}(min)$	kg S ha^{-1} yr^{-1}	Background rate of peat S accumulation	5
$S_{acc}(max)$	kg S ha^{-1} yr^{-1}	Maximum rate of peat S accumulation	16
S1	kg S ha^{-1} yr^{-1}	Total S deposition at which S accumulation increases above baseline rate	5
S2	kg S ha^{-1} yr^{-1}	Total S deposition at which S accumulation reaches maximum rate	20
DOC leach	kg C ha^{-1} yr^{-1}	DOC loss flux	210
DOC/DOS_{max}	g g^{-1}	Background ratio of DOC to DOS leaching	130
DOC/DOS_{min}	g g^{-1}	Minimum ratio of DOC to DOS leaching	60

Table 9.3: Site-specific parameters

Parameter	Units	Description	Source
Q	m/yr	Annual water flux from site	CL database
Cldep	meq/m2/yr	Annual mean Cl deposition	CBED
xBCdep	meq/m2/yr	Annual mean base cation deposition (may also include weathering BC inputs if necessary)	CBED

The procedure for running the peat acidity critical loads model is described in Appendix 4. The model has been applied, using a Python script, to the 1km distribution of the UK bog habitat currently in use for UK critical loads work (Hall et al, 2015a).

9.5 Model results

The CLmaxS map based on the current methodology was compared with the CLmaxS map resulting from running the new model (Figure 9.3); the mean CLmaxS value across the UK using the new method (1.32 keq ha^{-1} $year^{-1}$) was more than double the mean value using the current method (0.5 keq ha^{-1} $year^{-1}$). As the new method is focused on only generating CLmaxS, to compare exceedances using the old and new methods, exceedances have been calculated using non-marine sulphur deposition only, i.e.: Exceedance = sulphur deposition – ClmaxS. Using the latest CBED deposition data for 2012-14, resulted in 5.8% of the bog habitat area being exceeded with the old critical loads method, and no exceedance using the new method (Figure 9.4). As a further comparison, exceedances were also calculated using a FRAME hindcast scenario of sulphur deposition for 1970 when sulphur deposition was much higher; this resulted in almost 100% (99.3%) of the bog habitat being exceeded using the current CLmaxS data, and 26% using the new CLmaxS data (Figure 9.5).

The new model is currently parameterised assuming that all bogs are wet and in good condition, and therefore able to immobilise pollutants through peat formation. A degraded system where peat formation has ceased, will have little or no capacity to immobilise S or N, and may therefore be far more susceptible to acidification. Further work is required to (a) separately map the distribution of wet bogs in good condition and degraded bogs across the UK; (b) parameterise the model to calculate critical loads for degraded bogs.

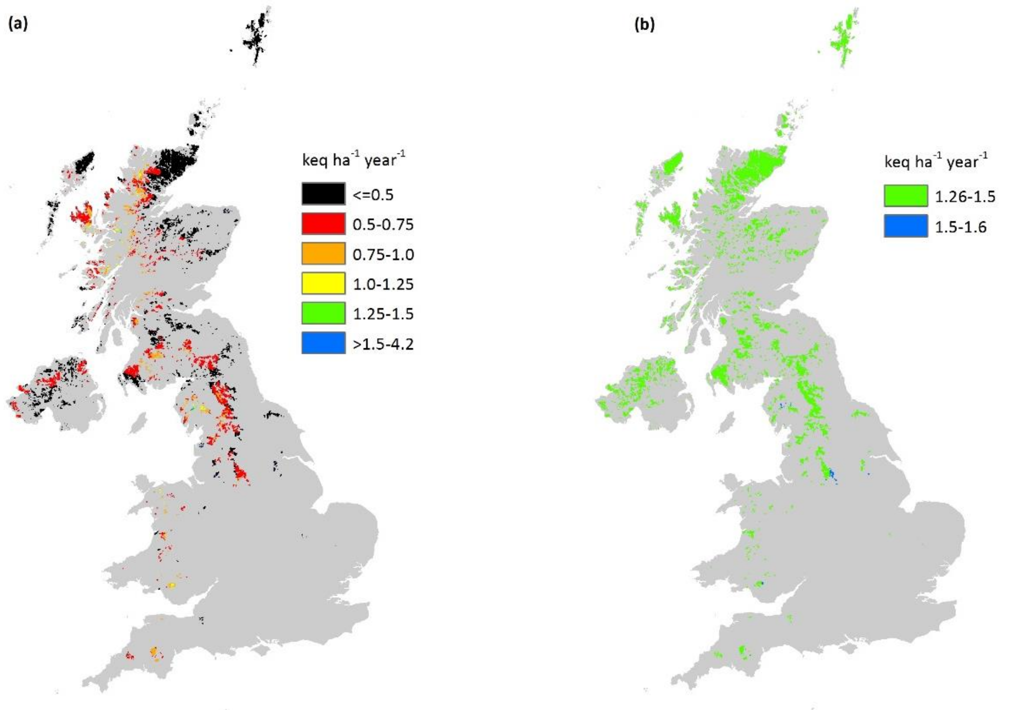


Figure 9.3: CLmaxS for peat soils in the UK based on (a) current methodology; (b) new methodology. Grey denotes areas of the UK dominated by non-peat soils.

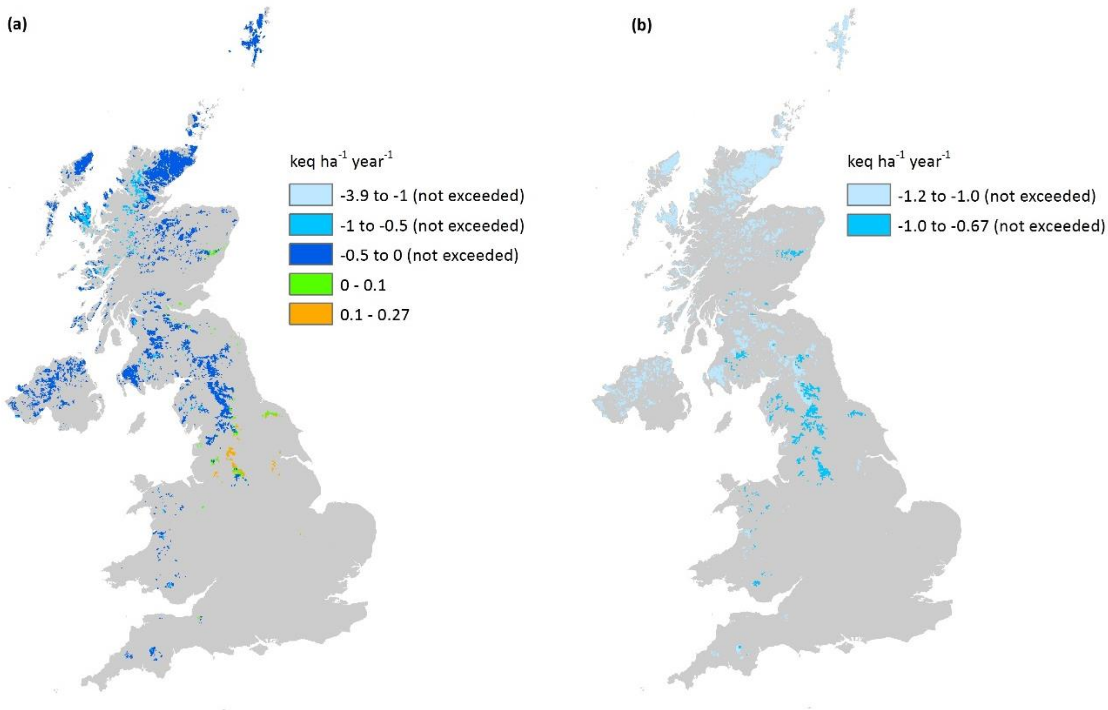


Figure 9.4: Exceedance of CLmaxS for peat soils by CBED sulphur deposition for 2012-14 for (a) critical loads based on the current methodology, and (b) the new methodology. Grey denotes areas of the UK dominated by non-peat soils.

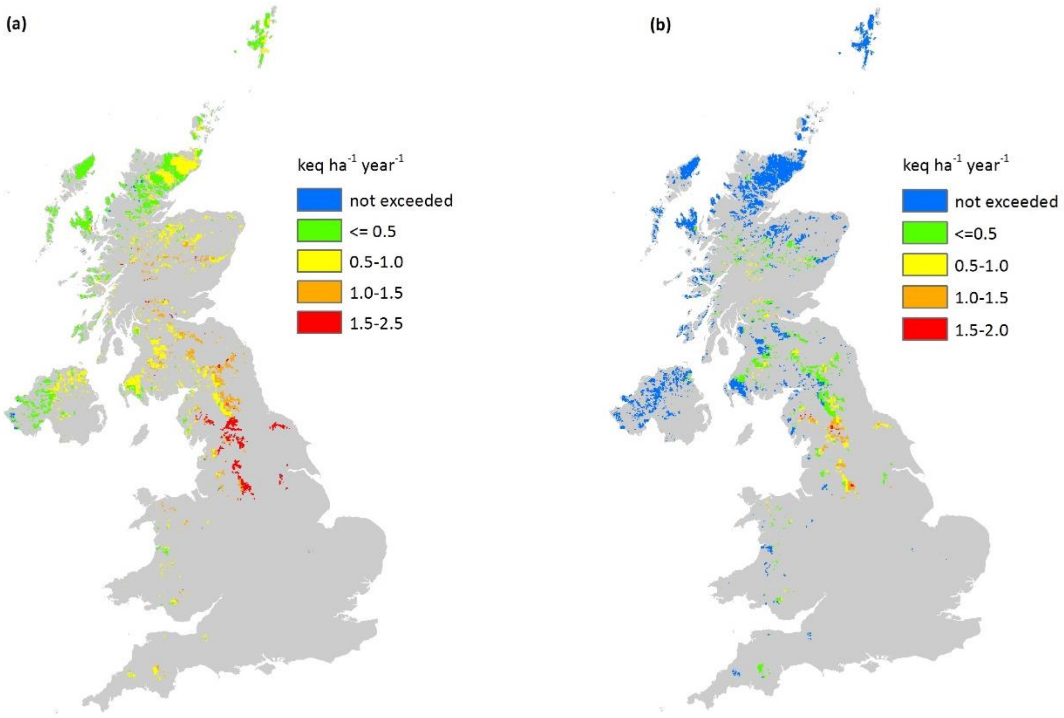


Figure 9.5: Exceedance of CLmaxS for peat soils by FRAME sulphur deposition for a hindcast scenario for 1970 for (a) critical loads based on the current methodology, and (b) the new methodology. Grey denotes areas of the UK dominated by non-peat soils.

10. Work Package 11: Provision of data and advice

Summary

- *The project provided advice and critical loads and deposition data to Defra, JNCC, Scottish Government, Welsh government, SEPA, Natural England, and Ricardo-AEA.*
- *The project carried out an ad-hoc study for Defra and JNCC to illustrate the ammonia deposition reductions that may be required in the future to meet protection targets for Annex I habitats.*
- *Members of the project team attended and gave presentations at annual meetings of the Committee on Air Pollution Effects Research (CAPER); a forum for informing policymakers and scientists on developments in the assessment of air pollution impacts on the natural and semi-natural environment.*

Section 10.1 summarises the activities for this WP under the contract to date; section 10.2 lists the range of other research projects that are, or have recently (in the last 2-3 years) been, using the UK critical loads data.

10.1 Provision of data and advice

Biodiversity Indicators

The trends in the areas of sensitive habitats with critical loads exceedance (see WP3) are used by Defra and JNCC as one of the UK's biodiversity indicators (http://jncc.defra.gov.uk/page-1824) and updated annually as requested for publication in "UK Biodiversity Indicators in Your Pocket" (BIYP). The summary exceedance statistics based on the CBED 2009-11 deposition data were provided to Defra and JNCC in May 2013 and the results for 2010-12 were submitted in May 2014. The text published with the indicator was sent to Jane Hall to check and update in October 2014. In addition, in November 2014, Jane Hall was contacted by Defra following a meeting of the Biodiversity Indicators Forum, to respond to a request for "statements of confidence for indicator assessments"; this was discussed with Defra and JNCC and the required information supplied.

As described in WP3, the exceedance trends were updated in 2015 using the revised CBED data from 2004 onwards. The updated trends covering the entire period from 1995 to 2013 and a short explanatory note outlining the reasons for the update were submitted to JNCC/Defra in August 2015 to update the BIYP indicators. This report additionally includes the exceedance results for 2012-14 which will be submitted during 2016 for updating the indicators, and added to the exceedance results on the project website.

Data to Scottish Government: Key Scottish Environment Statistics

Scottish Government were provided with the trends in exceedance statistics in June 2014, including results for 2010-12, for use in their "Key Scottish Environmental Statistics" publication for 2014. They were provided with the updated trends in critical load exceedances in August 2015.

Data to Welsh Government

Welsh Government (Environment and Social Justice Statistics) were provided with the trends in exceedance statistics in May 2014, including results for 2010-12. They were provided with the updated trends in critical load exceedances in August 2015.

Data to SEPA

SEPA have been provided with the summary SRCL exceedance statistics and exceedance maps for Scotland, based on the CBED 2009-11 deposition data, for use in a background paper for one of their management groups, and potentially for a "RoTAP" style report for Scotland.

Data and advice to JNCC

JNCC have been provided with (a) the 1km database of nutrient nitrogen critical loads for UK habitats together with the corresponding habitat areas; (b) SRCL for SACs, SPAs, SSSIs (c) exceedance of nutrient nitrogen SRCLs using CBED deposition for 2009-11. The nitrogen data will be used to generate maps of critical loads and exceedances for Northern Ireland in support of a strategic assessment of where new poultry farms could be placed. The SRCL data may be used to inform reporting on habitat/site status or condition.

Data and advice to Natural England

NE have been provided with (a) SRCL for SACs, SPAs, SSSIs; (b) CBED deposition for 2009-11; (c) exceedances of SRCL (SACs and SPAs) using CBED 2009-11 deposition data; (d) FRAME 2020 (UEP43 scenario) deposition data.

Data and advice to Ricardo-AEA

The current SRCL database (acidity and nitrogen), together with CBED deposition (2009-11) and FRAME 2020 (UEP43 scenario) have been provided to Ricardo-AEA for use in a Natural England project on the impacts of air pollution from roads on designated conservation sites.

Presentation to the Defra Air & Noise Group (October 2014)

Jane Hall gave a short presentation on critical loads and exceedances to the Defra Air & Noise Group, whilst visiting Defra for a project meeting in October 2014.

Presentations at Committee on Air Pollution Effects Research (CAPER) April 2015

Jane Hall gave a presentation on "The application of UK critical loads".
Ed Rowe gave a presentation on "Generating biodiversity-based critical load functions for nitrogen and sulphur using a dynamic habitat-suitability model.

Presentations at CAPER April 2016

Jane Hall gave a presentation on "Long-term trends in exceedance of critical loads for UK SACs".
Ed Rowe gave a presentation on "What use are predictions of biodiversity responses to air pollution?"

Protecting Annex I habitats post 2025

In 2013 Defra requested data analysis to calculate the ammonia deposition reductions required post 2025 to meet protection targets of <5% (ie, 95% protected) and <25% (ie, 75% protected) habitat area exceeded, for 41 Annex I habitats. The results of this study have been separately reported to JNCC and Defra (Appendix 3); this section provides a brief overview of the methods and results. The analysis was based on 41 selected Annex I habitats reported within UK SACs; recommended critical load values used by JNCC for Article 17 reporting under the Habitats Directive, were provided for habitats. Exceedances were calculated using FRAME 2025 deposition data (based on UEP45 emissions scenario), from which five separate deposition scenarios were derived (Table 10.1), in order to determine the habitat areas exceeded using different NHx deposition loads.

Table 10.1: Nitrogen deposition scenarios based on FRAME 2025 (UEP45) deposition data

Scenario	NOx deposition	NHx deposition
1	100%	0%
2	100%	25%
3	100%	50%
4	100%	75%
5	100%	100%

The results for each habitat were plotted to show the percentage area habitat exceeded against the NHx deposition budget for each scenario. From these plots, the percentage deposition reductions required to meet the protection targets were calculated. The results are summarised in Figure 10.1; this shows that nine habitats require 100% reduction in NHx deposition, plus reductions in NOx deposition to meet the 95% protection target, and this is still true for four of those habitats (alpine & boreal grassland, blanket bogs, sessile oak woods, yew woods) to meet the 75% protection target. In general the habitats requiring the largest reductions in NHx deposition are those with very low critical loads (eg, 5 kg N ha^{-1} $year^{-1}$), or woodlands (which receive higher deposition loads). The results are not surprising given that the NHx deposition predicted for moorland habitats in 2025 is > 10 kg N ha^{-1} $year^{-1}$ across most of the UK (and >20 kg N ha^{-1} $year^{-1}$ for deposition to woodland habitats).

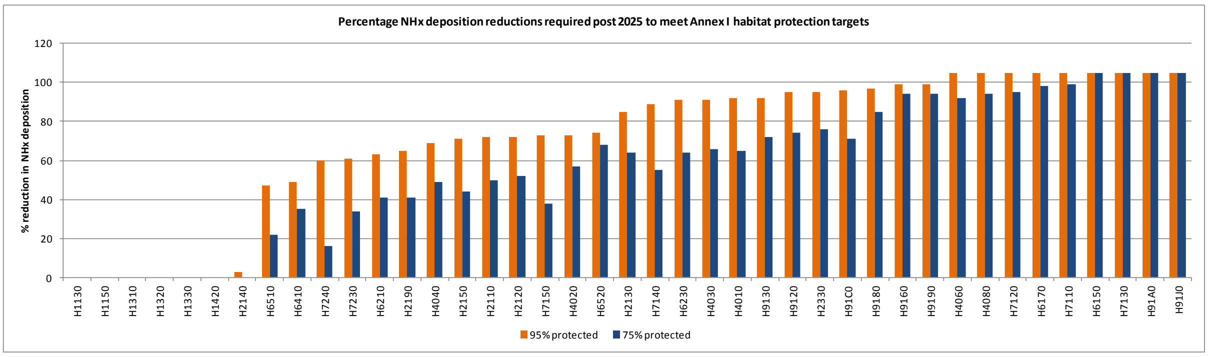


Figure 10.1: Plot showing the percentage reductions in NHx deposition required to meet the 75% and 95% protection targets for 41 Annex I habitats. Bars plotted above 100% represent those habitats where targets can only be met if all NHx deposition is removed AND reductions are also made in NOx deposition.

10.2 Use of UK critical loads data in other research projects

The UK critical loads data are utilised in an increasing number of other projects, mainly funded by Defra, or the SNCBs. The list below shows the current and recent projects using the UK values, with the funder in brackets:

- Atmospheric deposition at groundwater dependent wetlands (Environment Agency and British Geological Survey)
- Developing a decision framework to attribute nitrogen deposition as a threat to, or cause of, unfavourable habitat condition on protected sites (JNCC)

- ECLAIRE: Effects of Climate Change on Air Pollution and Response Strategies for European Ecosystems (EU)
- Air Pollution Information System (APIS)(Environment Agency, SNCBs, NIEA, SNIFFER, SEPA)
- Effusive Eruption Modelling project (Cabinet Office/Defra)
- Support for National Air Pollution Control Strategies (SNAPS)(Defra, AQ0947)
- Deposition Model Inter-comparison project (Defra, AQ0936)
- REBEND: Measures to evaluate the benefits to UK semi-natural habitats of reductions in nitrogen deposition (Defra, AQ0823)
- DivMet: A metric of nitrogen impacts on biodiversity for the UK response to a data request from the Coordination Centre for Effects (Defra, AQ0832)
- Identification of potential remedies for air pollution (nitrogen) impacts on designated sites (RAPIDS)(Defra, AQ0834)
- Air Quality Risk Assessment and SSSI Survey project (Natural England)

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APPENDICES

1: List of papers related to the ecological and functional changes in aquatic habitats in response to nitrogen deposition.

2: Draft minutes of the 2015 CCE Workshop and Task Force on Modelling and Mapping.

3: Report to Defra/AMEC on simulation of concentration and deposition of pollutants with future shipping scenarios and assessment of ecosystem and human health impacts.

4: Report to Defra on ammonia deposition reductions required to meet protection targets for Annex I habitats.

Appendix 1: List of papers related to the ecological and functional changes in aquatic habitats in response to nitrogen deposition (provided by Chris Curtis, ENSIS)

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Appendix 2: Report to Defra/AMEC on simulation of concentration and deposition of pollutants with future shipping scenarios and assessment of ecosystem and human health impacts.

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Emissions scenarios details to be reported by AMEC

The atmospheric chemical transport models FRAME and EMEP4UK were applied to simulate the future deposition of sulphur and nitrogen and concentrations of PM_{10} and $PM_{2.5}$ for the UK for three different scenarios of emissions of SO_2 and particulate matter from shipping. The results of the models were used to assess the impacts of the emissions abatements on natural ecosystems and on population weighted mean particulate concentrations.

Deposition modelling and ecosystem impact assessment

FRAME is a Lagrangian model using straight line trajectories with a 1^o angular resolution which runs at a 5 km resolution over the British Isles with a fine vertical grid spacing (1 m at the surface). Area emissions are injected into sector dependent levels and point source emissions are treated with a plume rise routine. Vertical diffusion in the air column is calculated using K-theory eddy diffusivity. Wet deposition is calculated using a 'constant drizzle' approximation driven by an annual rainfall map. Five land classes are considered and a vegetation specific canopy resistance parameterisation is employed to calculate dry deposition of SO_2, NO_2 and NH_3. The model chemistry includes gas phase and aqueous phase reactions of oxidised sulphur and oxidised nitrogen as well as aerosol formation. FRAME has been extensively applied to calculate sulphur deposition (Dore *et al.*, 2007) and nitrogen deposition (Dore *et al.*, 2012) over the UK as well as the exceedance of critical loads (Matejko et al., 2009). SNAP sector emissions scaling for predicted future emissions of SO_2, NO_x and NH_3 provided by Ricardo-AEA were used to define UK emissions for the year 2020. Three simulations were run with FRAME over the UK using different emissions of SO_2 from shipping as defined by the three scenarios.

The deposition of sulphur and oxidised and reduced nitrogen for Scenario 1 for the UK is illustrated in Table 1(a) and the sulphur deposition for all three scenarios is illustrated in Table 1(b). The reduction in SO_2 emissions for scenarios 2 and 3 resulted in decreases in total sulphur deposition to the UK of 6.0% and 10.3% respectively. The spatial distribution of the reductions in sulphur deposition between scenarios 1 and 3 is illustrated in figures 1(a) and 1(b). Dry sulphur deposition occurs mostly from SO_2 gas and the areas of greatest reductions occur close to the coast, particularly in the south-east. Wet deposition of sulphur occurs mostly due to the washout of sulphate aerosol particles which can be transported over long distances. The reduction in wet deposition of sulphur therefore occurs over a larger area of the country and is greatest in higher rainfall upland regions as well as the south of the country.

Table 1(a): Deposition of sulphur, oxidised and reduced nitrogen to the UK for scenario 1

Deposition (Gg S/N)	SO_x	NO_y	NH_x
Dry deposition	16.8	32.6	61.0
Wet deposition	41.1	44.3	63.1
Total deposition	57.9	76.9	124.1

Table 1(b): Deposition of sulphur to the UK for scenarios 1, 2 and 3 and % reduction in total sulphur deposition relative to scenario 1.

SOx Deposition (Gg S)	Scenario 1	Scenario 2	Scenario 3
Dry deposition	16.8	15.8	15.2
Wet deposition	41.1	38.6	36.7
Total deposition	57.9	54.4	51.9
% reduction	-	6.0	10.3

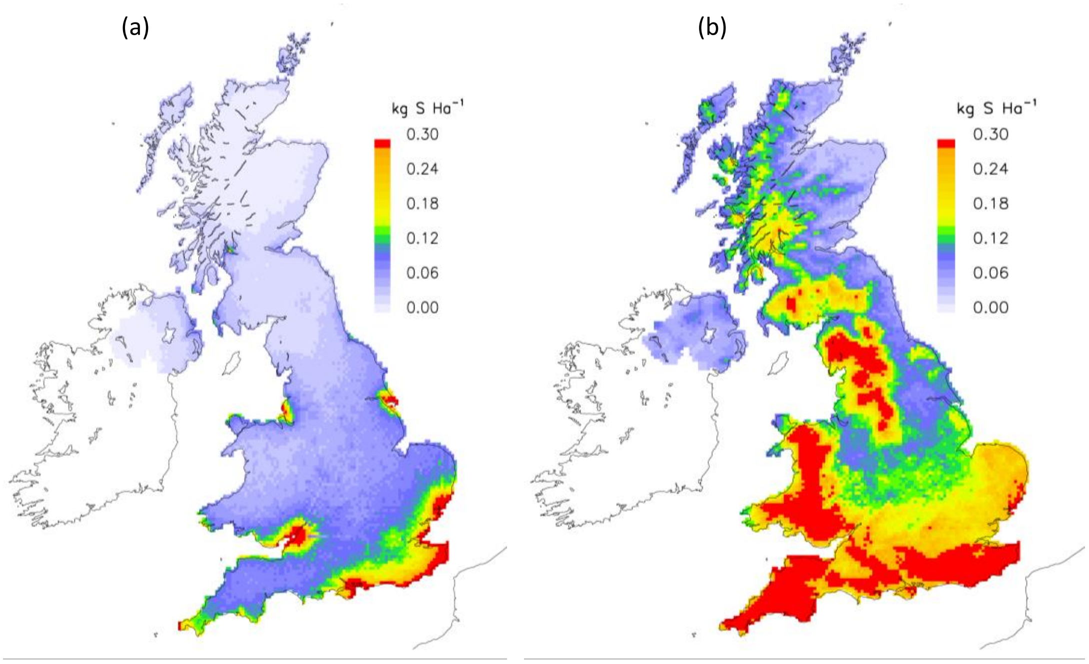

Figure 1: Reduction in: (a) dry deposition of sulphur and (b) wet deposition of sulphur between scenarios 1 and 3.

Critical loads exceedance statistics

Methods for the derivation of critical loads and the calculation of exceedances are described in detail in Hall et al (2014). Exceedance of acidity critical loads for UK habitats sensitive to acidification, have been calculated for all three scenarios and the results are summarised in Tables 2(a) and 2(b). The results show the differences in exceeded areas between the three scenarios are small, with scenario 2 reducing the area exceeded in the UK by 163km^2, and scenario 3 reducing the exceeded area by 460km^2 (a reduction in exceeded area of 0.6%), compared to scenario 1. The differences in the magnitude of exceedance, expressed as the Average Accumulated Exceedance (AAE), are also very small, at 0.01 keq ha^{-1} year^{-1}. In percentage terms the largest decrease in area exceeded (between scenarios 1 and 3) is for Wales (1.2%), which is consistent with the areas where sulphur deposition is shown to decline in Figure 1. In terms of area exceeded, the largest decrease is for Scotland (317km^2), most likely because this region has the largest area of sensitive habitats, but also the lowest deposition, and even a small decrease in deposition could result in exceeded areas becoming non-exceeded, whereas in other regions a larger decrease in deposition would be needed.

Table 2(a): Summary acidity critical load exceedance statistics by country for scenarios 1, 2 and 3.

Statistic	Scenario	England	Wales	Scotland	NI	UK
Habitat area (km^2)$^{\#}$	n/a	18635	7798	48083	3537	78051
Exceeded area (km^2)	1	11296	5464	11685	2435	30881
	2	11279	5444	11560	2435	30718
	3	11254	5372	11368	2424	30421
% area exceeded	1	60.6	70.1	24.3	68.8	39.6
	2	60.5	69.8	24.0	68.8	39.4
	3	60.4	68.9	23.6	68.5	39.0
AAE (keq ha^{-1} year^{-1})$^{\#\#}$	1	0.52	0.38	0.10	0.53	0.25
	2	0.52	0.38	0.10	0.53	0.24
	3	0.52	0.37	0.09	0.52	0.24

$^{\#}$The sum of the areas of habitats mapped as sensitive to acidification. It should be noted that the habitat distribution maps used for UK critical loads research (a) only include areas where data exist for the calculation or derivation of critical loads; (b) may differ from other national habitat distribution maps or estimates of habitat areas (Hall et al, 2014).

$^{\#\#}$AAE = Average Accumulated Exceedance calculated as:

$$\left(\sum(exceedance\ \times exceeded\ area)\right) \div total\ habitat\ area$$

where exceedance in keq ha^{-1} year^{-1} and areas in ha.

Table 2(b): Summary acidity critical load exceedance statistics for the UK by habitat for scenarios 1, 2 and 3.

Habitat	Habitat Area (km2)$^{\#}$	% area exceeded			AAE (keq ha^{-1} year^{-1})$^{\#\#}$		
		Scenario 1	Scenario 2	Scenario 3	Scenario 1	Scenario 2	Scenario 3
Acid grassland	15336	67.2	67.0	66.7	0.38	0.38	0.37
Calcareous grassland	1808	0.0	0.0	0.0	0.0	0.0	0.0
Dwarf shrub heath	24705	22.2	21.9	21.4	0.08	0.08	0.08
Bog	5454	42.4	42.1	41.8	0.26	0.25	0.25
Montane	3054	50.3	49.7	49.5	0.17	0.17	0.16
Managed conifer	8374	53.7	53.4	53.0	0.41	0.41	0.40
Managed broadleaf	7452	53.6	53.6	53.5	0.54	0.54	0.54
Unmanaged woodland	4011	40.9	40.8	40.7	0.36	0.36	0.36
Freshwaters$^{\#}$	7857	14.1	14.0	13.5	0.08	0.07	0.07
All habitats	78051	39.6	39.4	39.0	0.25	0.24	0.24

[#]The areas of UK habitats mapped as sensitive to acidification. It should be noted that the habitat distribution maps used for UK critical loads research (a) only include areas where data exist for the calculation or derivation of critical loads; (b) may differ from other national habitat distribution maps or estimates of habitat areas. The habitat area for freshwaters is based on the catchment areas of 1752 freshwater sites across the UK, mainly in upland and/or acid sensitive areas (Hall et al, 2014).
[##]AAE = Average Accumulated Exceedance calculated as:

$$\left(\sum(exceedance\ \times exceeded\ area)\right) \div total\ habitat\ area$$

where exceedance in keq ha^{-1} $year^{-1}$ and areas in ha.

Maps of AAE for all habitats combined have been created for each scenario; Figure 2 shows the map for scenario 1. The maps for scenarios 2 and 3 are almost identical to the map for scenario 1 and are not included in this report.

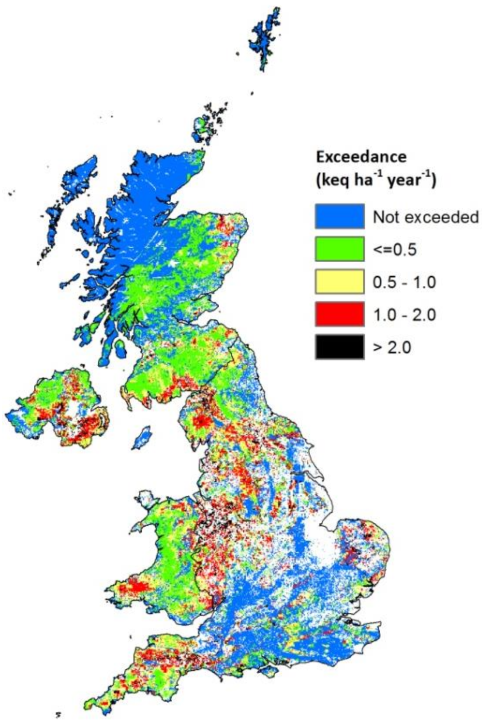

Figure 2: Average Accumulated Exceedance of acidity critical loads for all habitats combined by FRAME 2020 shipping scenario 1

Particulate concentration modelling and human exposure assessment

The EMEP4UK model is a Eulerian atmospheric chemical transport model driven by dynamic meteorology and can simulate surface ozone (Vieno *et al.*, 2010) and particulate concentrations (Vieno *et al.*, 2014) as well as deposition.

The impact on human health of implementing emissions reductions can be assessed by calculation of the Population Weighted Mean Concentrations (PWMC) of PM_{10} (particles with size less than 10 μm) and $PM_{2.5}$ (particles with size less than 2.5 μm). As well as emissions reductions in primary particulate matter, emissions reduction of SO_2 can also lead to a lowering in particulate concentrations due to a lower rate of formation of ammonium sulphate aerosol in the atmosphere. Emissions scaling factors for predicted future emissions provided by Ricardo-AEA for SO_2, NO_x, VOC, NH_3, CO, PM_{10} and $PM_{2.5}$ were used to define UK emissions for the year 2020.

The PWMC were calculated for the UK using spatially disaggregated population data for the UK supplied by Ricardo-AEA at a 1 km resolution. The results for the three scenarios are illustrated in Table 3. The PM concentrations calculated include all chemical components included in the EMEP4UK model. In the case of PM_{10} there is a significant contribution from sea salt. As the sea salt component is the same for all three scenarios, the effect of abatement of PM and SO_2 emissions from shipping is shown to result in a smaller % decrease in PM_{10} concentrations than that for $PM_{2.5}$. For scenario 3, a 1.4% reduction in PWMC for $PM_{2.5}$ was calculated relative to scenario 1.

Table 3: Population-weighted mean concentrations for the UK for PM_{10} and $PM_{2.5}$ concentrations for scenarios 1,2,3 ; Percentage reduction in population-weighted mean concentrations for scenarios 2 and 3 relative to scenario 1.

	Scenario 1	**Scenario 2**	**Scenario 3**
PM_{10}	15.23	15.15	15.10
$PM_{2.5}$	8.40	8.33	8.28
PM_{10} % reduction	-	0.48	0.83
$PM_{2.5}$ % reduction	-	0.79	1.41

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Appendix 3: Report to Defra on ammonia deposition reductions required to meet protection targets for Annex I habitats. (NB. Spreadsheet referred to not included in this Appendix)

Analysis of deposition levels required to reduce N impacts on conservation sites
[Jane Hall, 12/09/13; updated 23/10/13]

Methods

1) Site Relevant Critical Load tables set up in MS Access database for the SACs containing any one of the 41 Annex I Habitat Types listed in Table A1 of the "UK Approach to Assessing Conservation Status 2013 Article 17 Reporting" from JNCC, and using the critical load values defined in that table for each habitat.
2) FRAME 2025 5x5km deposition (calibrated to CBED 2009-2011) imported into database and five scenarios of N deposition created as in Table 1 below.

   Table 1: N deposition scenarios

Scenario	NOx deposition	NHx deposition
1	100%	0%
2	100%	25%
3	100%	50%
4	100%	75%
5	100%	100%

3) Deposition budgets calculated for each N deposition scenario.
4) FRAME deposition tables linked to SRCL tables to generate inputs for critical load exceedance calculations.
5) Exceedances calculated:
   - Assuming each Annex I habitat occurs everywhere within the SACs in which it is designated.
   - Using FRAME ecosystem-specific deposition (moorland, woodland, grid-average) values allocated to each Annex I habitat according to "Deposition Type" specified by JNCC for each habitat.
   - To give the area of habitat exceeded from all 1x1km squares or parts thereof that are exceeded within each site (ie, 5x5km deposition assumed to be constant across all 25 1x1km squares within each grid square, and exceedances calculated for each 1x1 km square or parts thereof for each site).
   - Results summarised by Annex I habitat by country and for the UK.
6) Exceedance results (% area exceeded) plotted against total N and NHx deposition budgets for each Annex I habitat, and results summarised.
7) Calculation of the NHx deposition budgets and NHx deposition reductions required to achieve the exceedance targets of 5% and 25% habitat area exceeded; hereafter referred to as the 95% and 75% habitat area protection targets.

Results

Refer to spreadsheet: CollatedResults_Defra_Sept2013.xlsx

(a) Deposition budgets: see sheet "DepBudgets"

This gives the budgets for grid-average, moorland, and woodland deposition for each of the five scenarios in Table 1 above.

(b) Annex I habitat areas in UK SACs: see sheet "Total areas"
Assuming all Annex I habitats occur everywhere within each SAC in which they are designated, for example, if a site is designated for 3 Annex I habitats, each habitat will be assigned the total area of the site. However, some of the Annex I habitats are coastal which extend beyond the UK land area and deposition data extent; the analysis has only been performed on those areas for which deposition data are available. The tables in this sheet show both the total area of Annex I habitats estimated from the areas of the SACs, and the areas used for the analysis; this shows that for the coastal habitats approximately 85% of the area is included in the analysis. This sheet also shows the nutrient nitrogen critical load assigned to each Annex I habitat and the ecosystem-specific deposition type assigned.

(c) Results by Annex I habitat: see individual sheets, one per habitat.
These give the following information:

- Habitat area (within SACs) by country – based on areas used for analysis in "Total areas" sheet.
- % habitat area exceeded by country for each scenario.
- Deposition budgets for deposition type allocated to the habitat. Deposition budgets for CBED 2004-06 and 2009-11 have been added to this sheet.
- The NHx deposition budget required to achieve (i) 95% habitat area protected; (ii) 75% habitat area protected; and the NHx deposition reductions required to achieve these targets based on linear interpolation of the exceedance results. For some habitats this linear interpolation may not be appropriate as there is a large step change from zero or very low exceedance with one scenario to 100% exceedance with the next highest deposition scenario.
- Plots of % habitat area exceeded by country against (i) total N depositon budgets for the UK; (ii) NHx deposition budgets for the UK.

The results for all Annex I habitats are summarised in seven tables below:
Tables 2a: shows the UK NHx deposition budgets and NHx deposition reductions required to achieve the targets of 95% and 75% habitat area protected. Note that for some of the more sensitive habitats with very low critical loads the area exceeded is >5% (and in some cases >25%) by NOx deposition alone. In these cases the NOx deposition budgets and reductions required to achieve the targets are given in addition to those needed for NHx.
Tables 2b-2e: provide the information as in Table 2a but for individual countries. Note that the deposition budgets and reductions are based on the totals for the UK (ie, not just the deposition to individual countries).
Table 2f: summarises the percentage reductions in NHx (and in some cases NOx) deposition required to achieve the targets for habitat protection.

Table 3 summarises the results by country to show which countries meet the 95% and 75% habitat area protection targets.

Table 2a: Summary results for the **UK**. Target UK NHx deposition calculated by linear interpolation from exceedance results for five NHx deposition scenarios.
(Target NOx deposition and NOx reductions required where habitat exceeded by >5% or >25% by NOx deposition alone)

Annex I habitat code	Annex 1 name	CLnutN(a) kg N ha^{-1} yr^{-1}	NOx deposition * kT N yr^{-1} (fixed)	NHx deposition* kT N yr^{-1}	Target NHx deposition to achieve 95% protection kT N yr^{-1}	NHx deposition reduction needed to meet 95% protection kT N yr^{-1}	Target NHx deposition to achieve 75% protection kT N yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N yr^{-1}
H1130^	Estuaries	30	96	400	#	#	#	#
H1150^	Coastal lagoons	30	110	211	NE	NE	NE	NE
H1310^	Salicornia on mud/sand	30	96	400	#	#	#	#
H1320^	Spartina swards	30	96	400	#	#	#	#
H1330^	Atlantic salt meadows	30	96	400	#	#	#	#
H1420^	Halophilous scrubs	30	96	400	NE	NE	NE	NE
H2110^	Embryonic shifting dunes	10	96	400	113	287	199	202
H2120^	Shifting dunes on shoreline	10	96	400	113	287	194	207
H2130^	Fixed dunes & herbaceous veg	8	96	400	61	339	144	256
H2140^	Decalcified fixed dunes	10	96	400	388	12	#	#
H2150^	Atlantic decalcified fixed dunes	10	96	400	115	285	223	178
H2190^	Humid dune slacks	10	96	400	140	260	234	166
H2330**	Inland dunes	8	96	400	20	381	98	302
H4010	Northern Atlantic wet heaths	10	96	400	33	367	138	262
H4020**	Temperate Atlantic wet heaths	10	96	400	109	291	171	230
H4030	European dry heaths	10	96	400	35	366	134	266
H4040**	Dry Atlantic coastal heaths	10	96	400	122	278	204	196
H4060	Alpine & boreal heaths	5	96	400	0 + (27)	400 + (69)	31	369
H4080	Sub-Arctic Salix spp. scrub	5	96	400	0 + (28)	400 + (68)	26	374
H6150	Siliceous alpine & boreal grassland	5	96	400	0 + (17)	400 + (79)	0 + (85)	400 + (11)
H6170	Alpine & subalpine calcareous grass	5	96	400	0 + (22)	400 + (74)	10	390
H6210	Semi-natural dry grassland	15	96	400	149	251	237	163
H6230	Species rich Nardus grassland	10	96	400	35	365	146	254
H6410	Molinia meadows	15	96	400	204	196	262	138
H6510	Lowland hay meadows	20	96	400	211	189	311	89
H6520**	Mountain hay meadows	10	96	400	103	297	129	271
H7110	Active raised bogs	5	96	400	0 + (22)	400 + (78)	5	395

Table 2a (**UK**) continued...

Annex I habitat code	Annex 1 name	CLnutN[(a)] kg N ha^{-1} yr^{-1}	NOx deposition * kT N ha^{-1} yr^{-1} (fixed)	NHx deposition* kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 95% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 5% protection kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 75% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N ha^{-1} yr^{-1}
H7120	Degraded raised bogs	5	96	400	0 + (57)	400 + (39)	22	378
H7130	Blanket bogs	5	96	400	0 + (160)	400 + (80)	0 + (78)	400 + (18)
H7140	Transition mires & quaking bogs	10	96	400	43	358	179	221
H7150	Depressions on peat substrates	10	96	400	109	291	247	154
H7230	Alkaline fens	15	96	400	155	245	265	136
H7240	Alpine pioneer formations	15	96	400	161	239	337	63
H9120**	Atlantic acidophilous beech forest	15	190	660	35	625	169	491
H9130**	Asperulo-Fagetum beech forest	15	190	660	54	607	184	476
H9160**	Oak or oak/hornbeam forests	15	190	660	8	652	41	619
H9180	Tilio-Acerion forests	15	190	660	20	640	100	560
H9190**	Old acidophilous oak woods	10	190	660	8	652	4	619
H91A0	Old sessile oak woods	10	190	660	0 + (21)	660 + (170)	0 + (103)	660 + (87)
H91C0	Caledonian forest	12	190	660	24	636	193	467
H91J0	Taxus baccata woods	5	190	660	0 + (10)	660 + (181)	0 + (48)	660 + (142)

^ Areas of coastal habitats may be under represented as deposition data exclude some coastal areas and these omitted from analysis.

[(a)] Critical load values as given in Table A1 of Annex 1 in “UKapproachconsultationv1.pdf” from JNCC.

NE = no exceedance

= exceeded area below threshold (5% or 25%)

* UK NHx deposition budgets for FRAME 2025 calibrated scenario; values for deposition type allocated to the habitat (by JNCC).

** Poor relationship between NHx deposition budget and percentage area exceeded; linear interpolation may not be appropriate.

Table 2b: Summary results for **England**. Target UK NHx deposition calculated by linear interpolation from exceedance results for five NHx deposition scenarios
(Target NOx deposition and NOx reductions required where habitat exceeded by >5% or >25% by NOx deposition alone)

Annex I habitat code	Annex 1 name	CLnutN[(a)] kg N ha^{-1} yr^{-1}	NOx deposition * kT N yr^{-1} (fixed)	NHx deposition* kT N yr^{-1}	Target NHx deposition to achieve 95% protection kT N yr^{-1}	NHx deposition reduction needed to meet 95% protection kT N yr^{-1}	Target NHx deposition to achieve 75% protection kT N yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N yr^{-1}
H1130^	Estuaries	30	96	400	#	#	#	#
H1150^	Coastal lagoons	30	110	211	NE	NE	NE	NE
H1310^	Salicornia on mud/sand	30	96	400	#	#	#	#
H1320^	Spartina swards	30	96	400	#	#	#	#
H1330^	Atlantic salt meadows	30	96	400	#	#	#	#
H1420^	Halophilous scrubs	30	96	400	NE	NE	NE	NE
H2110^	Embryonic shifting dunes	10	96	400	112	288	172	228
H2120^	Shifting dunes on shoreline	10	96	400	110	290	161	239
H2130^	Fixed dunes & herbaceous veg	8	96	400	35	365	122	278
H2140^	Decalcified fixed dunes	10	96	400	N/A	N/A	N/A	N/A
H2150^	Atlantic decalcified fixed dunes	10	96	400	128	272	215	186
H2190^	Humid dune slacks	10	96	400	127	274	213	187
H2330**	Inland dunes	8	96	400	20	381	98	302
H4010	Northern Atlantic wet heaths	10	96	400	13	387	68	332
H4020**	Temperate Atlantic wet heaths	10	96	400	109	291	171	230
H4030	European dry heaths	10	96	400	17	383	85	315
H4040**	Dry Atlantic coastal heaths	10	96	400	122	278	204	196
H4060	Alpine & boreal heaths	5	96	400	0 + (9)	400 + (87)	0 + (43)	400 + (53)
H4080	Sub-Arctic Salix spp. scrub	5	96	400	0 + (6)	400 + (89)	0 + (32)	400 + (64)
H6150	Siliceous alpine & boreal grassland	5	96	400	0 + (9)	400 + (87)	0 + (43)	400 + (53)
H6170	Alpine & subalpine calcareous grass	5	96	400	N/A	N/A	N/A	N/A
H6210	Semi-natural dry grassland	15	96	400	133	267	220	180
H6230	Species rich Nardus grassland	10	96	400	14	386	73	327
H6410	Molinia meadows	15	96	400	202	198	255	145
H6510	Lowland hay meadows	20	96	400	211	189	311	89
H6520**	Mountain hay meadows	10	96	400	103	297	128	272
H7110	Active raised bogs	5	96	400	0 + (12)	400 + (84)	0 + (59)	400 + (37)

Table 2b (**England**) continued...

Annex I habitat code	Annex 1 name	CLnutN[(a)] kg N ha^{-1} yr^{-1}	NOx deposition * kT N ha^{-1} yr^{-1} (fixed)	NHx deposition* kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 95% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 5% protection kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 75% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N ha^{-1} yr^{-1}
H7120	Degraded raised bogs	5	96	400	6	395	28	372
H7130	Blanket bogs	5	96	400	(7)	400 + (88)	(37)	400 + (59)
H7140	Transition mires & quaking bogs	10	96	400	11	389	54	346
H7150	Depressions on peat substrates	10	96	400	103	297	159	241
H7230	Alkaline fens	15	96	400	120	280	216	185
H7240	Alpine pioneer formations	15	96	400	81	319	210	190
H9120**	Atlantic acidophilous beech forest	15	190	660	36	624	127	533
H9130**	Asperulo-Fagetum beech forest	15	190	660	56	604	146	514
H9160**	Oak or oak/hornbeam forests	15	190	660	8	652	41	619
H9180	Tilio-Acerion forests	15	190	660	12	649	58	602
H9190**	Old acidophilous oak woods	10	190	660	8	652	41	619
H91A0	Old sessile oak woods	10	190	660	(15)	660 + (175)	(75)	660 + (115)
H91C0	Caledonian forest	12	190	660	N/A	N/A	N/A	N/A
H91J0	Taxus baccata woods	5	190	660	(10)	660 + (181)	(48)	660 + (143)

^ Areas of coastal habitats may be under represented as deposition data exclude some coastal areas and these omitted from analysis.

[(a)] Critical load values as given in Table A1 of Annex 1 in "UKapproachconsultationv1.pdf" from JNCC.

NE = no exceedance

= exceeded area below threshold (5% or 25%)

* UK NHx deposition budgets for FRAME 2025 calibrated scenario; values for deposition type allocated to the habitat (by JNCC).

** Poor relationship between NHx deposition budget and percentage area exceeded; linear interpolation may not be appropriate.

Table 2c: Summary results for **Wales**. Target UK NHx deposition calculated by linear interpolation from exceedance results for five NHx deposition scenarios
(Target NOx deposition and NOx reductions required where habitat exceeded by >5% or >25% by NOx deposition alone)

Annex I habitat code	Annex 1 name	CLnutN[(a)] kg N ha^{-1} yr^{-1}	NOx deposition * kT N yr^{-1} (fixed)	NHx deposition* kT N yr^{-1}	Target NHx deposition to achieve 95% protection kT N yr^{-1}	NHx deposition reduction needed to meet 95% protection kT N yr^{-1}	Target NHx deposition to achieve 75% protection kT N yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N yr^{-1}
H1130^	Estuaries	30	96	400	NE	NE	NE	NE
H1150^	Coastal lagoons	30	110	211	NE	NE	NE	NE
H1310^	Salicornia on mud/sand	30	96	400	NE	NE	NE	NE
H1320^	Spartina swards	30	96	400	NE	NE	NE	NE
H1330^	Atlantic salt meadows	30	96	400	NE	NE	NE	NE
H1420^	Halophilous scrubs	30	96	400	NE	NE	NE	NE
H2110^	Embryonic shifting dunes	10	96	400	176	224	239	161
H2120^	Shifting dunes on shoreline	10	96	400	192	208	243	157
H2130^	Fixed dunes & herbaceous veg	8	96	400	109	291	150	250
H2140^	Decalcified fixed dunes	10	96	400	N/A	N/A	N/A	N/A
H2150^	Atlantic decalcified fixed dunes	10	96	400	220	180	322	78
H2190^	Humid dune slacks	10	96	400	192	208	243	157
H2330**	Inland dunes	8	96	400	N/A	N/A	N/A	N/A
H4010	Northern Atlantic wet heaths	10	96	400	26	374	110	290
H4020**	Temperate Atlantic wet heaths	10	96	400	N/A	N/A	N/A	N/A
H4030	European dry heaths	10	96	400	30	370	115	285
H4040**	Dry Atlantic coastal heaths	10	96	400	N/A	N/A	N/A	N/A
H4060	Alpine & boreal heaths	5	96	400	(7)	400 + (89)	(35)	400 + (61)
H4080	Sub-Arctic Salix spp. scrub	5	96	400	N/A	N/A	N/A	N/A
H6150	Siliceous alpine & boreal grassland	5	96	400	(7)	400 + (89)	(32)	400 + (63)
H6170	Alpine & subalpine calcareous grass	5	96	400	(5)	400 + (90)	(27)	400 + (68)
H6210	Semi-natural dry grassland	15	96	400	206	194	232	168
H6230	Species rich Nardus grassland	10	96	400	11	389	54	346
H6410	Molinia meadows	15	96	400	212	188	270	130
H6510	Lowland hay meadows	20	96	400	N/A	N/A	N/A	N/A
H6520**	Mountain hay meadows	10	96	400	N/A	N/A	N/A	N/A
H7110	Active raised bogs	5	96	400	6	394	31	369

Table 2c (**Wales**) continued...

Annex I habitat code	Annex 1 name	CLnutN(a) kg N ha^{-1} yr^{-1}	NOx deposition * kT N ha^{-1} yr^{-1} (fixed)	NHx deposition* kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 95% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 5% protection kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 75% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N ha^{-1} yr^{-1}
H7120	Degraded raised bogs	5	96	400	(13)	400 + (83)	(65)	400 + (31)
H7130	Blanket bogs	5	96	400	(7)	400 + (88)	(37)	400 + (59)
H7140	Transition mires & quaking bogs	10	96	400	26	374	109	291
H7150	Depressions on peat substrates	10	96	400	22	378	103	297
H7230	Alkaline fens	15	96	400	149	251	227	173
H7240	Alpine pioneer formations	15	96	400	113	288	163	237
H9120**	Atlantic acidophilous beech forest	15	190	660	8	652	41	619
H9130**	Asperulo-Fagetum beech forest	15	190	660	31	629	154	506
H9160**	Oak or oak/hornbeam forests	15	190	660	N/A	N/A	N/A	N/A
H9180	Tilio-Acerion forests	15	190	660	13	647	67	593
H9190**	Old acidophilous oak woods	10	190	660	N/A	N/A	N/A	N/A
H91A0	Old sessile oak woods	10	190	660	(17)	660 + (173)	(83)	660 + (107)
H91C0	Caledonian forest	12	190	660	N/A	N/A	N/A	N/A
H91J0	Taxus baccata woods	5	190	660	(10)	660 + (180)	(49)	660 + (141)

^ Areas of coastal habitats may be under represented as deposition data exclude some coastal areas and these omitted from analysis.

(a) Critical load values as given in Table A1 of Annex 1 in "UKapproachconsultationv1.pdf" from JNCC.

NE = no exceedance

= exceeded area below threshold (5% or 25%)

* UK NHx deposition budgets for FRAME 2025 calibrated scenario; values for deposition type allocated to the habitat (by JNCC).

** Poor relationship between NHx deposition budget and percentage area exceeded; linear interpolation may not be appropriate.

Table 2d: Summary results for **Scotland**. Target UK NHx deposition calculated by linear interpolation from exceedance results for five NHx deposition scenarios

(Target NOx deposition and NOx reductions required where habitat exceeded by >5% or >25% by NOx deposition alone)

Annex I habitat code	Annex 1 name	CLnutN[(a)] kg N ha^{-1} yr^{-1}	NOx deposition * kT N yr^{-1} (fixed)	NHx deposition* kT N yr^{-1}	Target NHx deposition to achieve 95% protection kT N yr^{-1}	NHx deposition reduction needed to meet 95% protection kT N yr^{-1}	Target NHx deposition to achieve 75% protection kT N yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N yr^{-1}
H1130^	Estuaries	30	96	400	NE	NE	NE	NE
H1150^	Coastal lagoons	30	110	211	NE	NE	NE	NE
H1310^	Salicornia on mud/sand	30	96	400	NE	NE	NE	NE
H1320^	Spartina swards	30	96	400	NE	NE	NE	NE
H1330^	Atlantic salt meadows	30	96	400	NE	NE	NE	NE
H1420^	Halophilous scrubs	30	96	400	NE	NE	NE	NE
H2110^	Embryonic shifting dunes	10	96	400	207	193	313	87
H2120^	Shifting dunes on shoreline	10	96	400	229	171	376	24
H2130^	Fixed dunes & herbaceous veg	8	96	400	157	243	263	137
H2140^	Decalcified fixed dunes	10	96	400	388	12	#	#
H2150^	Atlantic decalcified fixed dunes	10	96	400	197	203	286	114
H2190^	Humid dune slacks	10	96	400	373	27	#	#
H2330**	Inland dunes	8	96	400	N/A	N/A	N/A	N/A
H4010	Northern Atlantic wet heaths	10	96	400	218	183	#	#
H4020**	Temperate Atlantic wet heaths	10	96	400	N/A	N/A	N/A	N/A
H4030	European dry heaths	10	96	400	207	193	#	#
H4040**	Dry Atlantic coastal heaths	10	96	400	N/A	N/A	N/A	N/A
H4060	Alpine & boreal heaths	5	96	400	(74)	400 + (21)	100	300
H4080	Sub-Arctic Salix spp. scrub	5	96	400	(50)	400 + (45)	51	349
H6150	Siliceous alpine & boreal grassland	5	96	400	(59)	400 + (37)	68	332
H6170	Alpine & subalpine calcareous grass	5	96	400	(37)	400 + (59)	32	368
H6210	Semi-natural dry grassland	15	96	400	N/A	N/A	N/A	N/A
H6230	Species rich Nardus grassland	10	96	400	213	197	#	#
H6410	Molinia meadows	15	96	400	307	93	335	65
H6510	Lowland hay meadows	20	96	400	N/A	N/A	N/A	N/A
H6520**	Mountain hay meadows	10	96	400	112	288	165	235
H7110	Active raised bogs	5	96	400	7	394	33	367

Table 2d (**Scotland**) continued...

Annex I habitat code	Annex 1 name	CLnutN[(a)] kg N ha^{-1} yr^{-1}	NOx deposition * kT N ha^{-1} yr^{-1} (fixed)	NHx deposition* kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 95% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 5% protection kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 75% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N ha^{-1} yr^{-1}
H7120	Degraded raised bogs	5	96	400	7	393	35	365
H7130	Blanket bogs	5	96	400	(85)	400 + (10)	103	297
H7140	Transition mires & quaking bogs	10	96	400	234	166	#	#
H7150	Depressions on peat substrates	10	96	400	237	163	#	#
H7230	Alkaline fens	15	96	400	351	49	#	#
H7240	Alpine pioneer formations	15	96	400	#	#	#	#
H9120**	Atlantic acidophilous beech forest	15	190	660	N/A	N/A	N/A	N/A
H9130**	Asperulo-Fagetum beech forest	15	190	660	N/A	N/A	N/A	N/A
H9160**	Oak or oak/hornbeam forests	15	190	660	N/A	N/A	N/A	N/A
H9180	Tilio-Acerion forests	15	190	660	264	396	608	52
H9190**	Old acidophilous oak woods	10	190	660	N/A	N/A	N/A	N/A
H91A0	Old sessile oak woods	10	190	660	40	620	310	350
H91C0	Caledonian forest	12	190	660	24	636	193	467
H91J0	Taxus baccata woods	5	190	660	N/A	N/A	N/A	N/A

^ Areas of coastal habitats may be under represented as deposition data exclude some coastal areas and these omitted from analysis.

[(a)] Critical load values as given in Table A1 of Annex 1 in “UKapproachconsultationv1.pdf” from JNCC.

NE = no exceedance

= exceeded area below threshold (5% or 25%)

* UK NHx deposition budgets for FRAME 2025 calibrated scenario; values for deposition type allocated to the habitat (by JNCC).

** Poor relationship between NHx deposition budget and percentage area exceeded; linear interpolation may not be appropriate.

Table 2e: Summary results for **Northern Ireland**. Target UK NHx deposition calculated by linear interpolation from exceedance results for five NHx deposition scenarios
(Target NOx deposition and NOx reductions required where habitat exceeded by >5% or >25% by NOx deposition alone)

Annex I habitat code	Annex 1 name	CLnutN[a] kg N ha^{-1} yr^{-1}	NOx deposition * kT N yr^{-1} (fixed)	NHx deposition* kT N yr^{-1}	Target NHx deposition to achieve 95% protection kT N yr^{-1}	NHx deposition reduction needed to meet 95% protection kT N yr^{-1}	Target NHx deposition to achieve 75% protection kT N yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N yr^{-1}
H1130^	Estuaries	30	96	400	NE	NE	NE	NE
H1150^	Coastal lagoons	30	110	211	NE	NE	NE	NE
H1310^	Salicornia on mud/sand	30	96	400	#	#	#	#
H1320^	Spartina swards	30	96	400	NE	NE	NE	NE
H1330^	Atlantic salt meadows	30	96	400	#	#	#	#
H1420^	Halophilous scrubs	30	96	400	NE	NE	NE	NE
H2110^	Embryonic shifting dunes	10	96	400	14	386	71	329
H2120^	Shifting dunes on shoreline	10	96	400	15	385	75	325
H2130^	Fixed dunes & herbaceous veg	8	96	400	41	359	116	284
H2140^	Decalcified fixed dunes	10	96	400	N/A	N/A	N/A	N/A
H2150^	Atlantic decalcified fixed dunes	10	96	400	11	389	55	345
H2190^	Humid dune slacks	10	96	400	208	192	263	137
H2330**	Inland dunes	8	96	400	N/A	N/A	N/A	N/A
H4010	Northern Atlantic wet heaths	10	96	400	44	357	124	276
H4020**	Temperate Atlantic wet heaths	10	96	400	N/A	N/A	N/A	N/A
H4030	European dry heaths	10	96	400	43	358	124	276
H4040**	Dry Atlantic coastal heaths	10	96	400	N/A	N/A	N/A	N/A
H4060	Alpine & boreal heaths	5	96	400	(21)	400 + (74)	5	395
H4080	Sub-Arctic Salix spp. scrub	5	96	400	N/A	N/A	N/A	N/A
H6150	Siliceous alpine & boreal grassland	5	96	400	(15)	400 + (80)	(77)	400 + (18)
H6170	Alpine & subalpine calcareous grass	5	96	400	105	295	125	275
H6210	Semi-natural dry grassland	15	96	400	N/A	N/A	N/A	N/A
H6230	Species rich Nardus grassland	10	96	400	207	194	232	168
H6410	Molinia meadows	15	96	400	324	77	#	#
H6510	Lowland hay meadows	20	96	400	N/A	N/A	N/A	N/A
H6520**	Mountain hay meadows	10	96	400	N/A	N/A	N/A	N/A
H7110	Active raised bogs	5	96	400	5	395	25	375

Table 2e (**Northern Ireland**) continued...

Annex I habitat code	Annex 1 name	CLnutN[(a)] kg N ha^{-1} yr^{-1}	NOx deposition * kT N ha^{-1} yr^{-1} (fixed)	NHx deposition* kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 95% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 5% protection kT N ha^{-1} yr^{-1}	Target NHx deposition to achieve 75% protection kT N ha^{-1} yr^{-1}	NHx deposition reduction needed to meet 75% protection kT N ha^{-1} yr^{-1}
H7120	Degraded raised bogs	5	96	400	5	395	25	375
H7130	Blanket bogs	5	96	400	(49)	400 + (47)	24	376
H7140	Transition mires & quaking bogs	10	96	400	108	293	141	259
H7150	Depressions on peat substrates	10	96	400	106	294	165	235
H7230	Alkaline fens	15	96	400	269	131	347	53
H7240	Alpine pioneer formations	15	96	400	N/A	N/A	N/A	N/A
H9120**	Atlantic acidophilous beech forest	15	190	660	N/A	N/A	N/A	N/A
H9130**	Asperulo-Fagetum beech forest	15	190	660	N/A	N/A	N/A	N/A
H9160**	Oak or oak/hornbeam forests	15	190	660	N/A	N/A	N/A	N/A
H9180	Tilio-Acerion forests	15	190	660	339	321	484	176
H9190**	Old acidophilous oak woods	10	190	660	N/A	N/A	N/A	N/A
H91A0	Old sessile oak woods	10	190	660	55	605	193	468
H91C0	Caledonian forest	12	190	660	N/A	N/A	N/A	N/A
H91J0	Taxus baccata woods	5	190	660	N/A	N/A	N/A	N/A

^ Areas of coastal habitats may be under represented as deposition data exclude some coastal areas and these omitted from analysis.

[(a)] Critical load values as given in Table A1 of Annex 1 in "UKapproachconsultationv1.pdf" from JNCC.

NE = no exceedance

= exceeded area below threshold (5% or 25%)

* UK NHx deposition budgets for FRAME 2025 calibrated scenario; values for deposition type allocated to the habitat (by JNCC).

** Poor relationship between NHx deposition budget and percentage area exceeded; linear interpolation may not be appropriate.

Table 2f. Percentage reductions in NHx deposition to UK required to meet target protection for Annex I habitats

(NOx deposition reductions required where targets cannot be achieved by reducing NHx deposition alone)

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	% UK deposition reduction required to meet 95% protection for:					% UK deposition reduction required to meet 75% protection for:				
			England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
H1130^	Estuaries	400	#	NE	NE	NE	#	#	NE	NE	NE	#
H1150^	Coastal lagoons	400	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE
H1310^	Salicornia on mud/sand	400	#	NE	NE	#	#	#	NE	NE	#	#
H1320^	Spartina swards	400	#	NE	NE	NE	#	#	NE	NE	NE	#
H1330^	Atlantic salt meadows	400	#	NE	NE	#	#	#	NE	NE	#	#
H1420^	Halophilous scrubs	400	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE
H2110^	Embryonic shifting dunes	400	72	56	48	96	72	57	40	22	82	50
H2120^	Shifting dunes on shoreline	400	72	52	43	96	72	60	39	6	81	52
H2130^	Fixed dunes & herbaceous veg	400	91	73	61	90	85	70	63	34	71	64
H2140^	Decalcified fixed dunes	400	N/A	N/A	3	N/A	3	N/A	N/A	#	N/A	#
H2150^	Atlantic decalcified fixed dunes	400	68	45	51	97	71	46	20	29	86	44
H2190^	Humid dune slacks	400	68	52	7	48	65	47	39	#	34	41
H2330**	Inland dunes	400	95	N/A	N/A	N/A	95	76	N/A	N/A	N/A	76
H4010	Northern Atlantic wet heaths	400	97	93	46	89	92	83	73	#	69	65
H4020**	Temperate Atlantic wet heaths	400	73	N/A	N/A	N/A	73	57	N/A	N/A	N/A	57
H4030	European dry heaths	400	96	93	48	89	91	79	71	#	69	66
H4040**	Dry Atlantic coastal heaths	400	69	N/A	N/A	N/A	69	49	N/A	N/A	N/A	49
H4060	Alpine & boreal heaths	400 (96)	100+(91)	100+ (93)	100+ (22)	100+ (78)	100+ (72)	100+(45)	100+(63)	75	99	92
H4080	Sub-Arctic Salix spp. scrub	400 (96)	100+(93)	N/A	100+ (47)	N/A	100+ (71)	100+(67)	N/A	87	N/A	94
H6150	Siliceous alpine & boreal grassland	400 (96)	100+(91)	100+ (93)	100+ (38)	100+ (84)	100+ (82)	100+(55)	100+(66)	83	100+(19)	100+(11)
H6170	Alpine & subalpine calcareous grass	400 (96)	N/A	100+ (94)	100+ (61)	74	100+ (77)	N/A	100+(71)	92	69	98
H6210	Semi-natural dry grassland	400	67	48	N/A	N/A	63	45	42	N/A	N/A	41
H6230	Species rich Nardus grassland	400	96	97	47	48	91	82	86	#	42	64
H6410	Molinia meadows	400	50	47	23	19	49	36	33	16	#	35
H6510	Lowland hay meadows	400	47	N/A	N/A	N/A	47	22	N/A	N/A	N/A	22
H6520**	Mountain hay meadows	400	74	N/A	72	N/A	74	68	N/A	59	N/A	68
H7110	Active raised bogs	400 (96)	100+ (88)	98	98	19	100+ (81)	100+(39)	92	92	94	99
H7120	Degraded raised bogs	400 (96)	99	100+ (86)	98	99	100+ (40)	93	100+(32)	91	94	95
H7130	Blanket bogs	400 (96)	100+ (92)	100+ (92)	100+ (11)	100+ (49)	100+ (84)	100+(61)	100+(61)	74	94	100+(19)
H7140	Transition mires & quaking bogs	400	97	94	42	73	89	86	73	#	65	55

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	% UK deposition reduction required to meet 95% protection for:					% UK deposition reduction required to meet 75% protection for:				
			England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
H7150	Depressions on peat substrates	400	74	95	41	73	73	60	74	#	59	38
H7230	Alkaline fens	400	70	63	12	33	61	46	43	#	13	34
H7240	Alpine pioneer formations	400	80	72	#	N/A	60	47	59	#	N/A	16
H9120**	Atlantic acidophilous beech forest	660	95	99	N/A	N/A	95	81	94	N/A	N/A	74
H9130**	Asperulo-Fagetum beech forest	660	92	95	N/A	N/A	92	78	77	N/A	N/A	72
H9160**	Oak or oak/hornbeam forests	660	99	N/A	N/A	N/A	99	94	N/A	N/A	N/A	94
H9180	Tilio-Acerion forests	660	98	98	60	49	97	91	90	8	27	85
H9190**	Old acidophilous oak woods	660	99	N/A	N/A	N/A	99	94	N/A	N/A	N/A	94
H91A0	Old sessile oak woods	660 (190)	100+(92)	100+(91)	94	92	100+(89)	100+(61)	100+(56)	53	71	100+(46)
H91C0	Caledonian forest	660	N/A	N/A	96	N/A	96	N/A	N/A	71	N/A	71
H91J0	Taxus baccata woods	660 (190)	100+(95)	100+(95)	N/A	N/A	100+(95)	100+(75)	100+(74)	N/A	N/A	100+(75)

^ Areas of coastal habitats may be under represented as deposition data exclude some coastal areas and these omitted from analysis.

[a] Critical load values as given in Table A1 of Annex 1 in “UKapproachconsultationv1.pdf” from JNCC.

NE = no exceedance

= exceeded area below threshold (5% or 25%)

* UK NHx deposition budgets for FRAME 2025 calibrated scenario; values for deposition type allocated to the habitat (by JNCC).

** Poor relationship between NHx deposition budget and percentage area exceeded; linear interpolation may not be appropriate.

Table 3: Summary results by country for each NHx deposition scenario (see table footnotes for more information)

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	Meets 95% protection for:					Meets 75% protection for:				
			England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
H1130	Estuaries	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		200	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		300	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		400	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
H1150^	Coastal lagoons	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		53	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		105	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		158	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		211	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
H1310^	Salicornia on mud/sand	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		200	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		300	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		400	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
H1320^	Spartina swards	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		200	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		300	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		400	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
H1330^	Atlantic salt meadows	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		200	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		300	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		400	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
H1420^	Halophilous scrubs	0	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		100	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		200	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		300	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		400	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓

Table 3 continued...

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	Meets 95% protection for: England	Wales	Scotland	NI	UK	Meets 75% protection for: England	Wales	Scotland	NI	UK
H2110	Embryonic shifting dunes	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	x	✓	✓	✓	✓	x	✓
		200	x	x	✓	x	x	x	✓	✓	x	x
		300	x	x	x	x	x	x	x	✓	x	x
		400	x	x	x	x	x	x	x	x	x	x
H2120	Shifting dunes on shoreline	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	x	✓	✓	✓	✓	x	✓
		200	x	x	✓	x	x	x	✓	✓	x	x
		300	x	x	x	x	x	x	x	✓	x	x
		400	x	x	x	x	x	x	x	x	x	x
H2130	Fixed dunes & herbaceous veg.	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	x	✓	✓	x	x	✓	✓	✓	✓	✓
		200	x	x	x	x	x	x	x	✓	x	x
		300	x	x	x	x	x	x	x	x	x	x
		400	x	x	x	x	x	x	x	x	x	x
H2140	Decalcified fixed dunes	0	N/A	N/A	✓	N/A	✓	N/A	N/A	✓	N/A	✓
		100	N/A	N/A	✓	N/A	✓	N/A	N/A	✓	N/A	✓
		200	N/A	N/A	✓	N/A	✓	N/A	N/A	✓	N/A	✓
		300	N/A	N/A	✓	N/A	✓	N/A	N/A	✓	N/A	✓
		400	N/A	N/A	x	N/A	x	N/A	N/A	✓	N/A	✓
H2150	Atlantic decalcified fixed dunes	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	x	✓	✓	✓	✓	x	✓
		200	x	✓	x	x	x	✓	✓	✓	x	✓
		300	x	x	x	x	x	x	✓	x	x	x
		400	x	x	x	x	x	x	x	x	x	x
H2190	Humid dune slacks	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		200	x	x	✓	✓	x	✓	✓	✓	✓	✓
		300	x	x	✓	x	x	x	x	✓	x	x
		400	x	x	x	x	x	x	x	✓	x	x

Table 3 continued...

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	Meets 95% protection for:					Meets 75% protection for:				
			England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
H2330**	Inland dunes	0	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		100	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		200	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		300	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		400	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
H4010	Northern Atlantic wet heaths	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	x	x	✓	x	x	x	✓	✓	✓	✓
		200	x	x	✓	x	x	x	x	✓	x	x
		300	x	x	x	x	x	x	x	✓	x	x
		400	x	x	x	x	x	x	x	✓	x	x
H4020**	Temperate Atlantic wet heaths	0	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		100	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		200	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		300	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		400	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
H4030	European dry heaths	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	x	x	✓	x	x	x	✓	✓	✓	✓
		200	x	x	✓	x	x	x	x	✓	x	x
		300	x	x	x	x	x	x	x	✓	x	x
		400	x	x	x	x	x	x	x	✓	x	x
H4040**	Dry Atlantic coastal heaths	0	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		100	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		200	x	N/A	N/A	N/A	x	✓	N/A	N/A	N/A	✓
		300	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		400	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
H4060	Alpine & boreal heaths	0	x	x	x	x	x	x	x	✓	✓	✓
		100	x	x	x	x	x	x	x	✓	x	x
		200	x	x	x	x	x	x	x	x	x	x
		300	x	x	x	x	x	x	x	x	x	x
		400	x	x	x	x	x	x	x	x	x	x

Table 3 continued...

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	Meets 95% protection for:					Meets 75% protection for:				
			England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
H4080	Sub-Arctic Salix spp. scrub	0	x	N/A	x	N/A	x	x	N/A	✓	N/A	✓
		100	x	N/A	x	N/A	x	x	N/A	x	N/A	x
		200	x	N/A	x	N/A	x	x	N/A	x	N/A	x
		300	x	N/A	x	N/A	x	x	N/A	x	N/A	x
		400	x	N/A	x	N/A	x	x	N/A	x	N/A	x
H6150	Siliceous alpine & boreal grassland	0	x	x	x	x	x	x	x	✓	x	x
		100	x	x	x	x	x	x	x	x	x	x
		200	x	x	x	x	x	x	x	x	x	x
		300	x	x	x	x	x	x	x	x	x	x
		400	x	x	x	x	x	x	x	x	x	x
H6170	Alpine & subalpine calcareous grass	0	N/A	x	x	✓	x	N/A	x	✓	✓	✓
		100	N/A	x	x	✓	x	N/A	x	x	✓	x
		200	N/A	x	x	x	x	N/A	x	x	x	x
		300	N/A	x	x	x	x	N/A	x	x	x	x
		400	N/A	x	x	x	x	N/A	x	x	x	x
H6210	Semi-natural dry grassland	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		200	x	✓	✓	✓	x	✓	✓	✓	✓	✓
		300	x	x	✓	✓	x	x	x	✓	✓	x
		400	x	x	✓	✓	x	x	x	✓	✓	x
H6230	Species rich Nardus grassland	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	x	x	✓	✓	x	x	x	✓	✓	✓
		200	x	x	✓	✓	x	x	x	✓	✓	x
		300	x	x	x	x	x	x	x	✓	x	x
		400	x	x	x	x	x	x	x	✓	x	x
H6410	Molinia meadows	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		200	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		300	x	x	✓	✓	x	x	x	✓	✓	x
		400	x	x	x	x	x	x	x	x	✓	x

Table 3 continued...

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	Meets 95% protection for:					Meets 75% protection for:				
			England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
H6510	Lowland hay meadows	0	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		100	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		200	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		300	x	N/A	N/A	N/A	x	✓	N/A	N/A	N/A	✓
		400	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
H6520**	Mountain hay meadows	0	✓	N/A	✓	N/A	✓	✓	N/A	✓	N/A	✓
		100	✓	N/A	✓	N/A	✓	✓	N/A	✓	N/A	✓
		200	x	N/A	x	N/A	x	x	N/A	x	N/A	x
		300	x	N/A	x	N/A	x	x	N/A	x	N/A	x
		400	x	N/A	x	N/A	x	x	N/A	x	N/A	x
H7110	Active raised bogs	0	x	✓	✓	✓	x	x	✓	✓	✓	✓
		100	x	x	x	x	x	x	x	x	x	x
		200	x	x	x	x	x	x	x	x	x	x
		300	x	x	x	x	x	x	x	x	x	x
		400	x	x	x	x	x	x	x	x	x	x
H7120	Degraded raised bogs	0	✓	x	✓	✓	x	✓	x	✓	✓	✓
		100	x	x	x	x	x	x	x	x	x	x
		200	x	x	x	x	x	x	x	x	x	x
		300	x	x	x	x	x	x	x	x	x	x
		400	x	x	x	x	x	x	x	x	x	x
H7130	Blanket bogs	0	x	x	x	x	x	x	x	✓	✓	x
		100	x	x	x	x	x	x	x	✓	x	x
		200	x	x	x	x	x	x	x	x	x	x
		300	x	x	x	x	x	x	x	x	x	x
		400	x	x	x	x	x	x	x	x	x	x
H7140	Transition mires & quaking bogs	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	x	x	✓	✓	x	x	✓	✓	✓	✓
		200	x	x	✓	x	x	x	x	✓	x	x
		300	x	x	x	x	x	x	x	✓	x	x
		400	x	x	x	x	x	x	x	✓	x	x

Table 3 continued...

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	Meets 95% protection for:					Meets 75% protection for:				
			England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
H7150	Depressions on peat substrates	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	x	✓	✓	✓	✓	✓	✓	✓	✓
		200	x	x	✓	x	x	x	x	✓	x	✓
		300	x	x	x	x	x	x	x	✓	x	x
		400	x	x	x	x	x	x	x	✓	x	x
H7230	Alkaline fens	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		100	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		200	x	x	✓	✓	x	✓	✓	✓	✓	✓
		300	x	x	✓	x	x	x	x	✓	✓	x
		400	x	x	x	x	x	x	x	✓	x	x
H7240	Alpine pioneer formations	0	✓	✓	✓	N/A	✓	✓	✓	✓	N/A	✓
		100	x	✓	✓	N/A	✓	✓	✓	✓	N/A	✓
		200	x	x	✓	N/A	x	✓	x	✓	N/A	✓
		300	x	x	✓	N/A	x	x	x	✓	N/A	✓
		400	x	x	✓	N/A	x	x	x	✓	N/A	x
H9120**	Atlantic acidophilous beech forest	0	✓	✓	N/A	N/A	✓	✓	✓	N/A	N/A	✓
		165	x	x	N/A	N/A	x	✓	x	N/A	N/A	✓
		330	x	x	N/A	N/A	x	x	x	N/A	N/A	x
		495	x	x	N/A	N/A	x	x	x	N/A	N/A	x
		660	x	x	N/A	N/A	x	x	x	N/A	N/A	x
H9130**	Asperulo-Fagetum beech forest	0	✓	✓	N/A	N/A	✓	✓	✓	N/A	N/A	✓
		165	x	x	N/A	N/A	x	✓	x	N/A	N/A	✓
		330	x	x	N/A	N/A	x	x	x	N/A	N/A	x
		495	x	x	N/A	N/A	x	x	x	N/A	N/A	x
		660	x	x	N/A	N/A	x	x	x	N/A	N/A	x
H9160**	Oak or oak/hornbeam forests	0	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		165	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		330	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		495	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		660	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x

Table 3 continued...

Annex I habitat	Annex I name	UK NHx budget* kT N/yr	Meets 95% protection for:					Meets 75% protection for:				
			England	Wales	Scotland	NI	UK	England	Wales	Scotland	NI	UK
H9180	Tilio-Acerion forests	0	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
		165	x	x	✓	✓	x	x	x	✓	✓	x
		330	x	x	x	✓	x	x	x	✓	✓	x
		495	x	x	x	x	x	x	x	✓	x	x
		660	x	x	x	x	x	x	x	x	x	x
H9190	Old acidophilous oak woods	0	✓	N/A	N/A	N/A	✓	✓	N/A	N/A	N/A	✓
		165	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		330	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		495	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
		660	x	N/A	N/A	N/A	x	x	N/A	N/A	N/A	x
H91A0	Old sessile oak woods	0	x	x	✓	✓	x	x	x	✓	✓	x
		165	x	x	x	x	x	x	x	✓	✓	x
		330	x	x	x	x	x	x	x	x	x	x
		495	x	x	x	x	x	x	x	x	x	x
		660	x	x	x	x	x	x	x	x	x	x
H91C0	Caledonian forest	0	N/A	N/A	✓	N/A	✓	N/A	N/A	✓	N/A	✓
		165	N/A	N/A	x	N/A	x	N/A	N/A	✓	N/A	✓
		330	N/A	N/A	x	N/A	x	N/A	N/A	x	N/A	x
		495	N/A	N/A	x	N/A	x	N/A	N/A	x	N/A	x
		660	N/A	N/A	x	N/A	x	N/A	N/A	x	N/A	x
H91J0	Taxus baccata woods	0	x	x	N/A	N/A	x	x	x	N/A	N/A	x
		165	x	x	N/A	N/A	x	x	x	N/A	N/A	x
		330	x	x	N/A	N/A	x	x	x	N/A	N/A	x
		495	x	x	N/A	N/A	x	x	x	N/A	N/A	x
		660	x	x	N/A	N/A	x	x	x	N/A	N/A	x

✓ = habitat meets protection target (area exceeded <= target percentage)
x = habitat does not meet protection target (habitat area exceeded > target percentage)
N/A = no habitat occurs/mapped for a country.
^ Areas of coastal habitats may be under represented as deposition data exclude some coastal areas and these omitted from analysis.
* UK NHx budgets for the five scenarios (0%, 25%, 50%, 75%, 100%) derived from the 2025 UEP45 scenario; values for deposition type allocated to the habitat (by JNCC).

Appendix 4: Procedure for running the new peat acidity critical loads model. (Refer to Section 9 of this report for further details; tables 9.2 and 9.3 copied below).

The model uses the input data listed in Tables 9.2 and 9.3, copied below as Tables 1 and 2. For implementation of the method at the UK-scale a Python script was written to perform the calculations.

Table 1: Fixed parameters used in the peat acidity critical loads model

Parameter	Units	Description	Default value
S_{acc}(min)	kg S ha^{-1} yr^{-1}	Background rate of peat S accumulation	5
S_{acc}(max)	kg S ha^{-1} yr^{-1}	Maximum rate of peat S accumulation	16
S1	kg S ha^{-1} yr^{-1}	Total S deposition at which S accumulation increases above baseline rate	5
S2	kg S ha^{-1} yr^{-1}	Total S deposition at which S accumulation reaches maximum rate	20
DOC leach	kg C ha^{-1} yr^{-1}	DOC loss flux	210
DOC/DOS_{max}	g g^{-1}	Background ratio of DOC to DOS leaching	130
DOC/DOS_{min}	g g^{-1}	Minimum ratio of DOC to DOS leaching	60

Table 2: Site-specific parameters

Parameter	Units	Description	Source
Q	m/yr	Annual water flux from site	CL database
Cldep	meq/m2/yr	Annual mean Cl deposition	CBED
xBCdep	meq/m2/yr	Annual mean base cation deposition (may also include weathering BC inputs if necessary)	CBED

First calculate the following:

(a) Convert chloride deposition to flux:
Cl = Cldep / Q

(b) Convert mean non-marine base cation deposition to flux:
xBC = xBCdep / Q

Then, repeat the following set of calculations with incrementally increasing values of total (non-marine plus marine) sulphur deposition (Sdep) until modelled acid neutralising capacity (ANC) is less than zero. The Sdep value at which this occurs is the critical load (i.e. CLmaxS, the maximum critical load of sulphur).

1. Calculate the rate of S accumulation in newly formed peat (set so that it can't exceed the total S deposition):
IF Sdep <= S1: Sacc = MIN{Sacc(min), Sdep}
IF Sdep >= S2: Sacc = Sacc(max)
IF S1 < Sdep < S2: Sacc = Sacc(min) + [{Sacc(max)-Sacc(min)}*(Sdep-S1)÷(S2-S1)]

2. Calculate DOS leaching, assuming this is proportional to the S content of newly formed peat:
   IF Sdep <= S1: $DOC/DOS = DOC/DOS_{max}$
   IF Sdep >=S2: $DOC/DOS = DOC/DOS_{min}$
   IF S1 < Sdep < S2: $DOC/DOS = DOC/DOS_{max} – [\{DOC/DOS_{max} – DOC/DOS_{min}\} * (Sdep – S1) \div (S2-S1)]$

   Then:
   DOSleach = MAX{MIN[DOCleach ÷ DOC/DOS),(Sdep-Sacc)},0}

3. Calculate sulphate leaching ($SO4_{leach}$) as a residual of Sdep minus sinks (set so that it can't fall below zero):
   $SO4_{leach}$ = MAX[(Sdep – Sacc – DOSleach), 0]

4. Calculate concentrations of porewater/runoff solute concentrations. Note that the calculation of ANC effectively factors out the marine component, which is neutral, and also (for CLmaxS) assumes no mineral N leaching:
   $SO4 = SO4_{leach} \div Q$
   xSO4 = SO4 – 0.104 * Cl
   ANC = xBC – xSO4

5. IF ANC < 0, CLmaxS = Sdep, otherwise repeat calculations 1-5