STATUS OF UK CRITICAL LOADS

CRITICAL LOADS METHODS, DATA & MAPS

FEBRUARY 2003

UK National Focal Centre, CEH Monks Wood, in collaboration with a range of UK experts

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Foreword by Defra

Aim

This Status Report documents the new updates to methods used to calculate and map critical loads for acidity and nutrient nitrogen for sensitive UK habitats, in light of new scientific findings. It is a factual scientific document, prepared by external experts. The Status report aims to summarise the changes in a transparent and detailed format to be helpful to stakeholders, policymakers and scientific experts.

The Critical Load as a policy tool

Air pollutants can adversely affect sensitive habitats, as highlighted in the report of the National Expert Group on Transboundary Air Pollution (2001). The critical load forms the basis of the effects-based approach, used to guide policy on reducing the environmental impacts of transboundary air pollutants, such as sulphur dioxide, nitrogen oxides and ammonia. The critical load is defined as:

'A quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge' (Nilsson & Grennfelt, 1988[1])

The amount of deposited pollutant above the critical load is termed the exceedance. Reducing the exceedance of critical loads is one of the main aims of international agreements to curb transboundary air pollution, such as the UNECE Protocol to Abate Acidification, Eutrophication and Ground-level ozone (1999), and the EC National Emission Ceilings Directive (2001). Both these agreements set national emission ceilings for sulphur dioxide, nitrogen oxides, ammonia and volatile organic compounds to be achieved from 2010. Both the Protocol and the Directive are scheduled for review in 2004/5, and critical loads will continue to be an important tool to evaluate potential future emission scenarios.

International Data on Critical Loads

Each party to the UNECE's Convention on Long Range Transboundary Air Pollution is required to submit spatial information on national critical loads and associated habitat areas, on a regular basis to the Coordination Centre on Effects (CCE) in the Netherlands. The CCE then combines the national data sets to produce critical load maps for Europe. These European maps are used to calculate exceedances for a range of potential European emission abatement scenarios, to be considered during negotiations. The most recent request from the CCE calls for data by the end of March 2003.

The UNECE's Mapping Manual (UBA, 1996) provides guidance on the calculation of critical loads, so that transparent common methods are applied across Europe. The Mapping Manual was last updated in 1996. Since that time, new research has provided further information on the adverse effects of air pollution on sensitive

[1] J. Nilsson and P. Grennfelt, 1988, Critical loads for sulphur and nitrogen. Report 1988:15. UNECE/Nordic Council of Ministers, Copenhagen, Denmark

habitats, particularly for nutrient nitrogen. In light of this, in November 2002, a UNECE workshop was held in Berne, to review and update the critical loads for nutrient nitrogen for a range of ecosystems. UK experts participated fully, and the results have been used in updating nutrient nitrogen critical loads for UK mapping (see below). The Mapping Manual is being updated to include the recommendations of the workshop.

The CCE requires habitat information to be submitted with critical load data. In order to harmonise the naming and classification of habitats used to calculate critical loads for Europe, the CCE now requires habitat codes from EUNIS (EUropean Nature Information System) for each of the habitat types for which critical loads are mapped.

UK Data on Critical Loads

The critical loads for UK habitats are calculated and mapped by the National Focal Centre (NFC) for Critical Loads, at CEH Monks Wood. The UK NFC website provides background information on critical loads, as well as status reports and updates, fully documenting the methods used to calculate critical loads for acidity and nutrient nitrogen. This Status Report (February 2003) is the latest in the series. In the last call for data from the CCE (February 2001), minor amendments were made to the critical load values. However, in recent months, the availability of new information, including the new CEH Land Cover Map 2000, has led to a thorough update of the national critical load data sets for acidity and nutrient nitrogen.

The UK's National Focal Centre produces maps at a resolution of 1km x 1km consistent with the resolution of national-scale soils data. The maps provide the broad pattern of sensitivity of habitats across the UK to acidity and nutrient nitrogen. However, they are not necessarily representative at the fine scale, where more detailed information would be required to produce a critical load relevant to a specific site.

Future developments on critical loads

Attention is now turning to the use of critical loads to gauge the potential impacts of air pollution at individual sites. Drivers such as the Habitats Directive and the Integrated Pollution, Prevention and Control Directive require an assessment of impacts on ecosystems at the site level. In close collaboration with the Countryside Council for Wales, English Nature, and the Joint Nature Conservation Committee, the Environment Agency are evaluating the application of the critical loads approach to ecologically relevant site-specific air pollution impact assessment. The updating of the national critical loads data set as described in this report, provides an important backdrop to the work being conducted at finer scale.

Next Steps

The NFC has also prepared maps of exceedance, by overlaying maps of deposition of acidity and nitrogen over the critical loads maps. These maps highlight the areas where critical loads are exceeded, and to what extent. As well as maps, statistics provide the areas of each habitat in England, Scotland, Wales and Northern Ireland exceeding critical loads. The maps and statistics are published on the critical loads web site (http://critloads.ceh.ac.uk) as an addendum to this report.

This report is compiled by the UK's National Focal Centre for Critical Loads, in collaboration with a range of experts, listed below.

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CONTENTS **PAGE**

1. EXECUTIVE SUMMARY

The air pollutants sulphur dioxide, nitrogen oxides and ammonia can contribute to acidification, and nitrogen oxides and ammonia can contribute to terrestrial eutrophication. Both problems can adversely affect semi-natural ecosystems. The common measure used across Europe since the 1980s, to gauge these problems, is the critical load. It is defined as 'a quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge'. Maps of critical loads assigned to the main ecosystems are produced for the UK on a grid at a resolution of 1km x 1km for terrestrial habitats, and 10km x 10km for freshwaters.

The preparation of critical loads maps has two main components (i) mapping the distribution of the main habitats and (ii) calculation of critical loads to assign to those habitats. Both these areas are the focus of considerable research, which is drawn together under the UK's National Focal Centre (NFC) for Critical Loads, at CEH Monks' Wood. The NFC's website (www.critloads.ceh.ac.uk) provides information on the methods used to calculate and map UK critical loads.

This document presents the latest update to the critical loads for acidity and nitrogen for sensitive UK habitats, which have been made in the light of new research findings and revisions to (i) the map of habitats, (ii) underlying data sets such as soil type, (iii) methods for calculating of critical loads. The updated data were submitted to the UNECE's Coordination Centre for Effects on 31 March 2003.

The main changes to the critical loads maps are summarised below. A selection of maps to illustrate the results of these changes is presented in Appendix 4, and are referenced in the main text.

Changes in habitats for which critical loads are mapped

Maps of land cover and species distributions are required to map critical loads for sensitive habitats. Previously, critical loads were mapped for six general ecosystem types: acid grassland, calcareous grassland, heathland, coniferous woodland, deciduous woodland and freshwaters, based upon the CEH Land Cover Map 1990 and additional data sets. The CEH Land Cover Map 2000 is now available, and has been used in this update, together with additional data on species distributions and other data sets, to map the critical loads for the terrestrial UK Biodiversity Action Plan Broad Habitats where both appropriate and feasible. Critical loads are now mapped for: calcareous grassland, acid grassland, dwarf shrub heath, bog, coniferous woodland, broad leaved, mixed and yew woodland, montane, and supralittoral sediment.

For freshwaters, critical loads are calculated using water chemistry samples taken from lakes and streams in acid-sensitive areas of the UK. From an original dataset of samples over 1500 sites, rigorous screening of the data in this update, has produced a set of 1163 lakes and streams, for which critical loads have been calculated and mapped on a 10km grid.

In order to harmonise the naming and classification of habitats across Europe, habitat codes from the EUNIS habitat classification scheme (Davies & Moss, 1999 and 2002) have also been provided for the first time for each of the habitat types for which critical loads are mapped.

Changes to underlying data sets

(i) National soil data (section 4.2)
The critical loads of acidity for soils are calculated for each 1km grid square based on the dominant soil type in that square. In the last year, the National Soil Resources Institute (NSRI, formerly the Soil Survey and Land Research Centre) has revised the 1km soils database for England and Wales, and the Macaulay Land Use Research Institute has updated and revised the 1km soils database for Scotland. These updates have resulted in changes to some of the percentage areas of the different soil types in each 1km grid square. As a consequence, the dominant soil series (or map unit), on which the empirical critical loads of acidity for soils are based, has changed in some grid squares leading to changes in the acidity critical loads map.

Changes in the national soils data have also led to revisions in the data sets of calcium weathering rates, base cation weathering rates, nitrogen immobilisation, denitrification and acidity critical loads for peat soils, which are all dependent on the dominant soil type.

(ii) National deposition data (Section 4.4.3)
The calculations of critical loads of acidity require non-marine base cation, non-marine chloride and total (marine plus non-marine) calcium deposition. These data sets have been updated to the measured values for 2000.

(iii) Individual input parameters (Section 4.4)
A number of specific input values (eg, nitrogen and base cation uptake, nitrogen leaching) have been updated based on the availability of new or additional data.

Changes in methods used to calculate critical loads (sections 4,7).

(i) Acidity critical loads for peat soils.
Changes in the number and distribution of peat-dominated 1km squares have resulted from the revision of the GB soils databases, including revisions to those soils classified as peats in Scotland. The method for the calculation of acidity critical loads for peat soils has been reviewed and a new method based on critical soil solution pH adopted.

(ii) Empirical critical loads of nutrient nitrogen.
In Berne in November 2002, a UNECE expert workshop reviewed the available literature and prepared a revised table of ranges of nutrient nitrogen critical loads for natural and semi-natural habitats, based on new scientific findings since 1996 (the date of the last update of the UNECE Mapping Manual). The critical loads for nutrient nitrogen used for UK mapping have been revised in light of the conclusions of the Berne workshop.

Recommendations for freshwaters

In the case of acidity critical loads for freshwaters, no changes to the method have been made in this update, however several parameters used to define nitrogen processes have been updated. The critical chemical criterion of ANC=0μeql^{-1} has not been updated. However, experts acknowledge that there is a growing body of evidence to suggest that the value of ANC=0μeql^{-1} currently used to calculate freshwater critical loads, does not provide adequate protection for freshwater biota. Further research is being conducted, and experts recommend a review of the ANC value as further evidence becomes available.

Next Steps – Exceedance calculations

The NFC has now used the updated critical load maps to calculate and map critical load exceedance, by comparison with national deposition maps for acidity and nitrogen. The resultant maps and statistics on the areas exceeding critical loads are published as an addendum to this report on the UK NFC web site (http://critloads.ceh.ac.uk).

2 AN INTRODUCTION TO CRITICAL LOADS

2.1 Introduction

The air pollutants sulphur dioxide, nitrogen oxides and ammonia can contribute to acidification, and nitrogen oxides and ammonia can contribute to terrestrial eutrophication. Both problems can adversely affect semi-natural ecosystems. The National Expert Group on Transboundary Air Pollution recently reviewed the impacts of air pollutants on UK ecosystems and prospects for the future (NEGTAP, 2001). Measuring and quantifying the potential ecological damage by air pollutants is not a simple matter. The common measure, used across Europe since the 1980s, is the critical load. This is defined as **'**a quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge' (Nilsson & Grennfelt, 1988).

The amount of deposited pollutant that exceeds the critical load of acidity or nutrient nitrogen, is called the 'exceedance'. Exceedance of critical loads represents the potential for damage, but is not a quantitative estimate of damage to the environment. The critical load is an equilibrium concept and gives no information on the timescales for damage (when the critical load is exceeded) or recovery (when deposition is reduced below the critical load). Timescales for damage and recovery vary greatly, depending on the environmental receptor and the pollutant combination; to estimate these dynamic models are required. Such models are being developed under Defra contracts and under the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP), but are not discussed in this Report.

The application of critical loads has proved very useful for policy development. It provides an 'effects-based' approach where the environmental benefits of emission reductions can be gauged. Closing the gap between estimated pollutant deposition (above the critical load) and the critical load, for ecosystems across Europe, is one of the main drivers of emission control agreements under the UNECE CLRTAP, the EC European Acidification Strategy, and Clean Air For Europe.

2.2 Calculation and Mapping of Critical Loads

The preparation of Critical Loads maps has two main components: (i) mapping the distribution of the main habitats and (ii) calculation of critical loads to assign to those habitats. This report documents the updates made to the habitats maps and critical load values in light of new scientific findings.

Maps of the main terrestrial habitats, on a 1km x 1km grid, are generated using the CEH Land Cover Map 2000 and additional data such as species distributions. In this update, Biodiversity Action Plan Broad Habitats are used for the calculation of critical loads for the first time. In addition, in order to harmonise the naming and classification of habitats across Europe, habitat codes from the EUNIS habitat classification scheme (Davies & Moss, 1999 and 2002) are also provided for each of the habitat types for which critical loads are mapped.

A number of methods exist to determine the critical loads of acidity or nutrient nitrogen, which fall into two broad categories (i) mass balance and (ii) empirical approaches. In the mass balance approach, the long term chemical inputs and outputs (affecting acidity or nitrogen) are calculated, and the critical load is exceeded when the critical chemical criterion is breached. The chemical criterion is chosen to reflect a change in the ecosystem which would lead to damage. In the empirical approach, the critical load is estimated rather than calculated, based on experimental and field evidence for the ecosystem response to deposition.

Appropriate methods, critical chemical criteria and ranges for empirical critical loads are agreed at the UNECE level under the International Cooperative Programme on Modelling and Mapping. These methods are summarised in the UNECE's Mapping Manual (http://www.umweltbundesamt.de/mapping). The methods currently used in the UK to calculate acidity and nutrient nitrogen critical loads are consistent with the Mapping Manual and are summarised in Table 2.1. Appendix 1 provides an overview of the UK methods to calculate critical loads for (A) terrestrial habitats and (B) freshwaters. Appendix 1 also describes the Critical Load Function (CLF) used to examine the effects of sulphur and nitrogen deposition which can simultaneously contribute to acidification.

Table 2.1 Summary of the methods used to calculate critical loads for sensitive habitats in the UK

Habitat type	**Method to assign critical load for acidity**	**Method to assign critical load for nutrient nitrogen**
Unmanaged woodland	Steady State Mass Balance	Empirical
Managed woodland	Steady State Mass Balance	Steady State Mass Balance
Non-woodland terrestrial habitats	Empirical, based on dominant soil type	Empirical
Freshwater lakes and streams	First Order Acidity Balance [FAB]	Not used[1]

[1] Acidified freshwaters in UK are assumed not to be susceptible to eutrophication, due to phosphorus limitation
Further information on methods in Appendix 1

Research on the ecological effects of acidification and eutrophication continues in the UK and Europe. As new findings emerge, it is necessary to update the critical load values on a regular basis. Changes in critical load values can emerge as a result of

a) changes in the underlying data sets used to calculate critical loads, e.g. soil maps
b) changes in the effects criterion used to determine damage, e.g. threshold value of ANC (Acid Neutralising Capacity) for freshwaters
c) changes in the methodology to calculate critical loads, e.g. calculation of acidity critical loads for peats.

This report documents the updates made to the habitats maps and critical load values in light of new scientific findings.

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J. Nilsson and P. Grennfelt, 1988, Critical loads for sulphur and nitrogen. Report 1988:15. UNECE/Nordic Council of Ministers, Copenhagen, Denmark

3. HABITAT MAPPING

3.1 Introduction

Critical loads are mapped for habitats sensitive to acidification and/or eutrophication. Therefore information on the location and distribution of these habitats is required to enable them to be mapped. Updates to these habitat maps have been made based on the new Land Cover Map 2000 (Fuller *et al.*, 2002(a) & 2002(b)), species distribution data (Preston, Pearman & Dines, 2002), vegetation classification information (NVC; Rodwell, 1991-2000) and revised 1km soil maps.

In previous years, critical loads have been calculated and mapped for six ecosystems in the UK: acid grassland, calcareous grassland, heathland, coniferous woodland, deciduous woodland and freshwaters (Hall *et al.*, 1998; Hall *et al.*, 2001). The areas of terrestrial ecosystems were primarily defined from the 1990 CEH Land Cover Map (LCM1990) of Great Britain (Fuller *et al*., 1994), and in some cases the areas were refined using species distribution data. For Northern Ireland the level 3 CORINE land cover map was translated into classes of the LCM1990. The freshwater ecosystem areas were defined by the catchment boundaries of the lakes and streams sampled by University College London under contract to the then Department of the Environment.

3.2 Updated maps - Broad Habitat Critical Load Maps (2003)

3.2.1 Land Cover Map 2000

In 2002, the CEH Land Cover Map 2000 (LCM2000) was completed (Fuller *et al.*, 2002(a) & 2002(b)). Both LCM1990 and LCM2000 were derived using classified satellite imagery. The 2000 map includes Northern Ireland and uses a new classification enabling many of the UK Biodiversity Action Plan (BAP) Broad Habitats to be identified and mapped. The habitat categories in LCM2000 relevant to this work are listed in Appendix 2. In agreement with Defra, in this February 2003 update, the NFC has calculated and mapped critical loads for selected BAP Broad Habitats, enabling both critical loads and their exceedances to be considered in relation to these habitats of conservation importance. No changes have been made to the catchment boundaries of the lakes and streams, however, a number of freshwater sites have been removed from the critical loads database using several screening criteria.

Figure 3.1 in Appendix 4 shows LCM1990 and LCM2000.

3.2.2 Refinements

The LCM2000 identified 16 target classes (level 1), which are sub-divided into 27 sub-classes (level 2) to allow the construction of the widespread Broad Habitats (Appendix 2). However, there are limitations in using satellite data to map very specific habitat types. Therefore, the NFC in collaboration with LCM2000 and habitats experts, has developed a method to refine the LCM2000 using additional data sets (such as species distribution, altitude and soil type), to map terrestrial Broad Habitats for this work.

To produce the Broad Habitat Critical Load Maps for acid grassland, calcareous grassland, dwarf shrub heath and bog, maps of species distributions have been used to refine the LCM2000 data. Preston *et al.* (2003) identified all species associated with individual BAP Broad Habitats, and produced 10km resolution maps showing the percentage of species in each 10km square, making adjustments for the latitudinal gradient in species diversity in the UK.[2] In collaboration with habitat experts, a cut-off value for the percentage of species that best represent the key areas for the habitats has been applied. For calcareous grassland, the cut-off value is 50% (i.e. 10km squares where more than 50% of the species pool is present have been selected). In all other cases, a cut-off of 40% has been used.

The 10km squares selected using the species distribution data were overlaid on the corresponding 1km LCM map, and the 1km LCM squares falling within the 10km squares were mapped to represent the habitat. In some cases, additional data have also been used to sub-divide the habitats. For example, the 1km Hydrology of Soil Types (Boorman *et al.*, 1995) data were used to distinguish between wet and dry areas of acid grassland and of dwarf shrub heath.

For the coniferous and broadleaved woodland habitats, a combination of LCM and Forest Research data have been used to distinguish the managed and unmanaged woodland areas.

The montane broad habitat (represented by *Racomitrium* heath) and the Atlantic oak woods (a sub-habitat of unmanaged broadleaved and coniferous woodland) required more specific information to map them. Therefore a combination of LCM2000 data, 10km mapped classes of the National Vegetation Classification (Rodwell, 1991(a), 1991(b), 1992, 1995, 2000) and for montane, altitude data, have been used.

Further information on the combinations of data used to map the individual habitats is given later in Sections 3.7.1 – 3.7.6.

It is pointed out that in all cases, the LCM2000 is used throughout as the base map for terrestrial habitats, since this provides the habitat area values for each 1km square. These areas are used in assessments of national critical load exceedance and are also required by the CCE. Whilst the additional data (10km maps of species distributions or National Vegetation Classification classes) are useful to refine the habitat distribution maps, they cannot be used alone, since they do not provide the habitat area values at the required resolution.

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[2] Preston *et al.* (2003) used habitat associations of vascular plants, based on field quadrat data to calculate the frequency of plant species within the BAP Broad Habitat types. Two major sources of quadrat data were used: (i) the original data used to derive the National Vegetation Classification (Rodwell, 1991(a), 1991(b), 1992, 1995, 2000); (ii) quadrat samples collected by Countryside Survey 2000 (Haines-Young *et al.*, 2000). The table of frequencies from these datasets was used to calculate preference indices for species to broad habitat categories. Species diversity in a 10km square was defined simply as the number of species for each habitat type that were recorded for the square. Species distribution data were derived from the *New Atlas* of plants (Preston *et al.*, 2002); records prior to 1930 were excluded. The species diversity in a 10km square was then compared to the species diversity of its biogeographic zone to account for the latitudinal gradient in species diversity within the UK.

3.3 Assigning Critical Loads to Broad Habitats

As in previous years, for acidity, the empirical critical loads for soils are applied to areas of non-woodland terrestrial habitats. These critical loads are set to protect the soils upon which the habitats depend. This contrasts with the mass balance acidity critical loads for woodland habitats and the nutrient nitrogen (both empirical and mass balance) critical loads, which aim to protect both the soils and the vegetation. This is explained in more detail in Chapter 4. The methods for assigning critical loads to freshwaters remain unchanged (Chapter 6).

3.4 Harmonising habitat classification at UNECE CLRTAP level

It is useful for UK policy purposes to map critical loads for the BAP Broad Habitats, identified in the UK Biodiversity Action Plan. However, different habitats may be more appropriate in other countries. This leads to critical loads being assigned to a wide range of habitat types across Europe. In order to improve transparency at the UNECE CLRTAP level, in 2000, the UK NFC carried out a study as a "contribution in kind" to the International Cooperative Programme on Modelling and Mapping (ICPMM) to harmonise the definitions of ecosystems for which countries submitted critical loads data (Hall, 2001).

The UK NFC's study showed that countries identify and map their sensitive ecosystems from a variety of data sources, ranging from land cover maps to aerial photographs or survey data. However, no information had previously been collated on the methods and data used to define these ecosystems; countries submitted an ecosystem name with their critical loads data to the CCE and it was assumed that all ecosystems with the same name represented similar ecosystems. A method to harmonise the definitions of ecosystems was required to enable comparisons of critical loads data and maps for similar ecosystems in different countries.

The UK NFC study recommended the EUropean Nature Information System (EUNIS)(Davies & Moss, 1999 & 2002). EUNIS is a hierarchical habitat classification scheme, developed for pan-European applications (not specifically for critical loads) and enables habitats to be assigned a habitat code regardless of the source of data used for mapping them. The key advantage of EUNIS to critical loads work is that it provides a consistent method of habitat classification across Europe enabling cross-comparison of results on a habitats basis. EUNIS has been adopted by the CCE and ICPMM and countries are now asked by the CCE to submit the appropriate EUNIS habitat codes with their critical loads data. In line with this approach, the recent UNECE expert workshop on empirical nutrient nitrogen critical loads (Bobbink *et al.*, in press) used the EUNIS classification as a basis for setting critical load values for sensitive habitats.

3.5 Assigning EUNIS codes to UK BAP Broad Habitats

For the UK, although national mapping activities are now focused on the BAP Broad Habitats, the data submitted to the CCE need to have the relevant EUNIS habitat codes assigned. The BAP Broad Habitats and EUNIS systems identify and name ecosystems using different methods. Empirical critical loads for nutrient nitrogen have been agreed at the UNECE level using EUNIS codes to identify the habitats

(Chapter 7). The UK NFC has therefore identified the corresponding BAP Broad Habitat, so that UK critical loads for nutrient nitrogen can be consistently mapped in terms of Broad Habitats. Conversely, all other critical loads (for acidity and mass balance nutrient nitrogen) have been mapped on the Broad Habitat level, and the UK NFC has identified the corresponding EUNIS classes. The relationships between the BAP Broad Habitats, EUNIS classes and the habitats mapped are given in Appendix 3. ***However, it should be noted that there is rarely a direct relationship between the BAP Broad Habitats and the EUNIS classes; the two schemes are not directly interchangeable.***

3.6 Overview of uncertainties

The Broad Habitat Critical Load maps have been produced using the best available data and have been discussed and agreed by habitat experts. Although *they may not include every small area of each sensitive habitat at the regional or local scale, they do give* ***national*** *pictures of the main habitat types*, adequate for ***national*** critical loads mapping purposes.

There are however, uncertainties associated with the maps. The main reasons are:

- There are uncertainties in all the data sets used (land cover, forest land use data, species distributions, NVC classes, soils data, altitude data)
- The Broad Habitat Critical Load Maps are presented at a resolution of 1km, for consistency with the critical loads data, however, they are based on a combination of data sets at different resolutions (e.g. 1km land cover and 10km species distributions).
- Where the 10km species distribution maps are used to refine habitat areas from the LCM2000, the 10km grid squares selected represent the Broad Habitat in terms of the species composition present (above the percentage threshold used). However, this does not necessarily mean that all the species occur within every 1km grid square within each 10km square; the habitat area could therefore be overestimated.
- The 10km NVC class maps have the same uncertainties associated with them as the 10km species data above.

3.7 Specific mapping information for each Broad Habitat

The paragraphs below describe the methods used to map each habitat in this Update, highlighting any differences between the methods for mapping the habitat for acidity critical loads, and the more specific habitats mapped for nutrient nitrogen critical loads.

3.7.1 Woodland habitats

The UK BAP identifies two woodland Broad Habitats: “broadleaved, mixed and yew woodland” and “coniferous woodland”. For critical loads both managed and unmanaged woodlands are included, since the long-term protection of the whole ecosystem function (ie, soils, trees, linked aquatic ecosystems) is important. However, these managed and unmanaged systems are treated separately as the critical loads are determined by different approaches. While LCM2000 distinguishes between broadleaved and coniferous woodland, satellite imagery cannot be used alone to separate managed from unmanaged woodland, or to identify specific types of woodland, such as Atlantic oak woods. Therefore, a combination of LCM2000 data, Forest Research (FR) data and National Vegetation Classification (NVC) data have been used in the mapping of these habitats. The FR data consisted of a combination of the National Inventory of Woodland and Trees (NIWT) and the Ancient and Semi-natural Woodland Inventories of English Nature, the Countryside Council for Wales and Scottish Natural Heritage (FC, 2001; FC, 2002a; FC 2002b). Together these data identified areas of managed coniferous woodland, managed broadleaved woodland, unmanaged coniferous and broadleaved woodland and Atlantic oak woods. The unmanaged woodland consists of ancient and semi-natural woodland, yew and Scots Pine and is "managed" for biodiversity or amenity, but not timber production. All other coniferous and broadleaved woodland is assumed to be primarily managed as productive forest where harvesting and removal of trees takes place.

For this update, four classes of woodland within the BAP woodland Broad Habitats have been mapped:

- managed (productive) coniferous woodland
- managed (productive) broadleaved woodland
- unmanaged (ancient & semi-natural) coniferous and broadleaved woodland
- Atlantic oak woods (a sub-division of the unmanaged woodland)

Each of these is presented as a separate map because it is possible for more than one woodland type to occur in a 1km grid square (see below).

Critical loads for nutrient nitrogen are assigned to each of the above woodland types using different methods (Chapter 7). Critical loads for acidity are calculated using the Simple Mass Balance method, using different values of calcium, base cation and nitrogen uptake for managed and unmanaged woodlands (Chapter 4).

For consistency with the mapping of other habitats, the LCM2000 data provides the basis for the woodland habitat areas. The LCM woodland data were compared with the FR data; although the two sets of data coincide in many areas, there is not a complete match for a number of reasons:

- The data sets have been generated using different methods and for different purposes.
- LCM is a map of land cover, whereas the FR data are for land use.
- Unlike FR data, LCM does not distinguish between the managed and unmanaged woodland areas.
- FR data can include other habitat types, for example, areas of young trees that would be classified as non-woodland cover types (eg, grassland, heathland) on the LCM.

To overcome these differences, a method has been developed in agreement with FR, that uses the ratio of the different FR woodland types in each 1km square to estimate the areas of woodland from the LCM data (see below). The paragraphs below describe the methods used to map the four woodland types.

Managed (productive) coniferous woodland (Figure 3.2)

First the FR 1km data for managed coniferous woodland were overlaid onto the LCM class (2.1) for coniferous woodland. Then the distribution of managed coniferous woodland was mapped as those 1km grid squares where both FR and LCM data occur. The managed coniferous woodland areas were calculated as:

Managed conifers = (ratio of FR managed coniferous woodland area to FR total woodland area) * LCM coniferous woodland area

where FR total woodland area = sum of managed and unmanaged coniferous and broadleaved woodland.

Managed (productive) broadleaved woodland (Figure 3.3)
First the FR 1km data for managed broadleaved woodland were overlaid onto the LCM class (1.1) for broadleaved/mixed woodland. Then the distribution of managed broadleaved woodland was mapped as those 1km grid squares where both FR and LCM data occur. The managed broadleaved woodland areas were calculated as:

Managed broadleaved = (ratio of FR managed broadleaved woodland area to FR total woodland area) * LCM broadleaved/mixed woodland area

Unmanaged (ancient & semi-natural) coniferous & broadleaved woodland (Figure 3.4)
First the FR data for unmanaged broadleaved and coniferous woodland were overlaid onto the LCM total woodland (ie, sum of LCM classes 1.1 and 2.1). Then the distribution of unmanaged woodland was mapped as those 1km grid squares where both FR and LCM data occur. Areas mapped as Atlantic oak woods (see below) form a sub-set of the unmanaged woodland area and were therefore removed from this map. The remaining unmanaged woodland areas were calculated as:

Unmanaged woodland = (ratio of FR unmanaged area to FR total woodland area) * LCM total woodland area

Atlantic oak woods (Figure 3.5)
The first stages in generating this map are identical to generating the unmanaged woodland map. The FR data for unmanaged broadleaved and coniferous woodland were overlaid onto the LCM total woodland (ie, sum of LCM classes 1.1 and 2.1) and the distribution of unmanaged woodland mapped as the 1km grid squares where both FR and LCM data occur. This map was then overlaid with the 10km grid squares of the National Vegetation Classification woodland class W17 (*Quercus petraea – Betula pubescens – Dicranum majus* woodland). The 1km unmanaged woodland areas within the 10km squares were selected to represent areas of Atlantic oak woods. The areas of these woods were calculated in the same way as the unmanaged woodland above.

Woodland areas for Northern Ireland
The LCM2000 includes areas of coniferous and deciduous woodland for NI. However, data are not currently available for this region to distinguish managed from unmanaged woodland. The Environment and Heritage Service (David Mitchel, EHS, pers. comm.) advise that (a) all the coniferous woodland in NI would be managed (b) the majority of broadleaved woodland is semi-natural woodland with only a small percentage of broadleaved plantation; the latter is not necessarily managed, as a large proportion of this is estate amenity woodland. In addition, the NVC data for the

woodland class W17 are also unavailable for NI at the present time. Therefore, based on the information from EHS, the following approach has been used in this update to map two types of woodland for NI:

- Managed coniferous woodland: mapped using LCM class 2.1 (coniferous woodland) only and assuming all areas managed.
- Unmanaged broadleaved woodland: mapped using LCM class 1.1 (broadleaved/mixed woodland) only and assuming all areas unmanaged.

Summary of the use of the different woodland habitat maps

Critical loads have been calculated for the four woodland classes mapped. However, as the FR data used for GB do not distinguish between unmanaged coniferous woodland and unmanaged broadleaved woodland, these woodlands cannot be mapped separately or correctly associated with their respective BAP Broad Habitats. Hence, for this update, the unmanaged coniferous woodland (including native Scots Pine) has been included with unmanaged broadleaved woodland under the BAP Broad Habitat for "Broadleaved, mixed and yew woodland". The critical loads data have been submitted to the CCE by the EUNIS codes given in the tables below.

BAP Broad Habitat 1: Broadleaved, mixed and yew woodland

Critical load	EUNIS class	Woodland habitats mapped
Acidity	G1 Broadleaved woodland*	Managed (productive) broadleaved woodland
	G1&G3 Broadleaved & coniferous woodland*	(i) Unmanaged (ancient & semi-natural) coniferous & broadleaved woodland (ii) Atlantic oak woods
Nutrient nitrogen	G1 Broadleaved woodland**	Managed (productive) broadleaved woodland
	G1&G3-GF Broadleaved & coniferous woodland (effects on ground flora only)***	Unmanaged (ancient & semi-natural) coniferous & broadleaved woodland
	G1-LA Broadleaved woodland (effects on epiphytic lichens only)***	Atlantic oak woods

* Simple Mass balance for acidity method used
** Steady State mass balance for nutrient nitrogen method used
*** Empirical critical loads used

BAP Broad Habitat 2: Coniferous woodland

Critical load	EUNIS class	Woodland habitats mapped
Acidity	G3 Coniferous woodland*	Managed (productive) coniferous woodland
Nutrient nitrogen	G3 Coniferous woodland**	Managed (productive) coniferous woodland

* Simple Mass balance for acidity method used
** Steady State mass balance for nutrient nitrogen method used

3.7.2 Grassland habitats

Only two of the BAP grassland Broad Habitats are mapped for critical loads: acid grassland and calcareous grassland. However, it is not possible to distinguish these grassland habitats using satellite imagery alone. Hence a three-class "soil acid sensitivity" map (Hornung *et al*., 1995), based on soil pH and base saturation, was used in combination with the original grassland imagery in LCM 2000 to produce three separate acid, neutral and calcareous LCM grassland classes (Table 3.1 below; Fuller *et al.,* 2002(a) & 2002(b)).

This method worked reasonably well for defining the acid grassland areas. The calcareous grassland areas may be overestimated as the "soil acid sensitivity" class (pH >5.5) used is likely to include some areas of grassland with a more neutral pH. However, the calcareous grassland map obtained using this method shows a reasonable correspondence with the species data for this habitat. Acid and calcareous grassland are therefore included in the Broad Habitat Critical Loads maps. Species data have been used to refine the distributions (see below). However, neutral grassland is excluded, and critical loads for acidity and nutrient nitrogen for neutral grassland are not mapped for three reasons:

- The pH range of the "soil acid sensitivity" map class used for this grassland type (Table 3.1) tends towards the acid side of neutral, so areas of neutral grassland are likely to be overestimated. Species data for neutral grassland do not help in this case since they cover many areas where grassland does not appear on LCM 2000.
- As explained in Section 7.3.1, a critical load for nutrient nitrogen has not been assigned to neutral grassland; it is therefore not necessary to map the habitat.
- Neutral grassland in the UK is largely composed of improved grasslands, including hay meadows.

Table 3.1. Definition of the three classes of the "soil acid sensitivity" map by Hornung et al (1995) and their use in LCM 2000

Soil acid sensitivity class	Base saturation	pH	LCM grass category
Highly sensitive	<20%	<4.5	Acid grassland
Moderately sensitive	20-60%	>4.5 and < 5.5	Neutral grassland
Low sensitivity	>60%	>5.5	Calcareous grassland

BAP Broad Habitat 7: Calcareous grassland

Two maps of calcareous grassland have been generated: one for the areas sensitive to acidification and the other for areas sensitive to eutrophication. For nutrient nitrogen critical loads the 1km LCM map of calcareous grassland was overlaid with the 10km species data for this Broad Habitat, and the 1km LCM areas within the 10km squares selected for mapping (Figure 3.6).

Some of the 1km calcareous grassland squares mapped for nutrient nitrogen critical loads coincide with 1km squares that have low empirical soil acidity critical loads (ie, below 2.0 keq ha^{-1} $year^{-1}$). The soil acidity critical loads are based on the dominant soil type in each square (see later, Section 4.2); soils derived from base-poor rocks are more acid and result in low critical loads. Calcareous grassland may occur in 1km squares that have a low soil acidity critical load, but is unlikely to be found on the acid soil determining the low soil critical load. The soils upon which the calcareous grassland occurs are likely to have a higher acidity critical load. Therefore, when mapping acidity critical loads for calcareous grassland nationally, squares with an empirical soil acidity critical load below 2.0 keq ha^{-1} $year^{-1}$ are omitted from the map, on the basis that the critical load (calculated using the empirical method based on the dominant soil) is not appropriate for this grassland soil (Figure 3.7).

It should be noted that the threshold for excluding soil acidity critical loads squares has been amended in this update from 0.5 keq ha^{-1} $year^{-1}$ to 2.0 keq ha^{-1} $year^{-1}$.

BAP Broad Habitat 7: Calcareous grassland

Critical load	EUNIS class	Habitat map used
Acidity	E1.26 Sub-Atlantic semi-dry calcareous grassland	Combination of LCM & species data and excluding areas where empirical soil acidity critical loads < 2.0 keq ha^{-1} $year^{-1}$
Nutrient nitrogen	E1.26 Sub-Atlantic semi-dry calcareous grassland	Combination of LCM & species data

BAP Broad Habitat 8: Acid grassland (Figures 3.8 and 3.9)

To provide the map for acidity critical loads the LCM 2000 acid grassland class was overlaid with the 10km species data for the habitat, and the 1km LCM areas within the 10km squares selected. For nutrient nitrogen the areas of acid grassland needed to be separated into areas of wet and dry grassland to represent and map the critical loads for two EUNIS classes (see below). Although the UK mapping value for both the wet and dry grassland is the same (Chapter 7), it is necessary to identify the separate areas for submitting data to the CCE by EUNIS class. The 29 classes of the 1km Hydrology of Soil Types (HOST) map were divided into wet and dry categories (Table 3.2). The HOST class for each 1km grid square is based on the dominant soil type in the square, so each square can only be defined as having either wet or dry soils. These 1km data have been overlaid on the acid grassland map defined above, enabling wet and dry grassland to be mapped separately. However, for the UK maps (Figures 3.8 and 3.9) these have been combined into a single “acid grassland” map, since only wet or dry grassland can be mapped in any 1km grid square.

Critical load	EUNIS class	Habitat maps used
Acidity	E1.7 Non-Mediterranean dry acid and neutral closed grassland* E3.5 Moist or wet oligotrophic grassland	Combination of LCM and species data
Nutrient nitrogen	E3.5 Moist or wet oligotrophic grassland	Combination of LCM and species data within areas of wet soils
	E1.7 Non-Mediterranean dry acid and neutral closed grassland*[#]	Combination of LCM and species data within areas of dry soils

* Although the definition of EUNIS class E1.7 includes both acid and neutral grassland, only acid grassland is mapped in this update.
# The empirical nutrient nitrogen critical loads assigned to EUNIS class E1.7 are based on evidence for acid grasslands only (see Section 7.3.1).

3.7.3 Heathland habitats

The BAP dwarf shrub heath Broad Habitat map is based on the LCM2000 classes for both dwarf shrub heath and open shrub heath. The habitat area is further refined by selecting the LCM areas within the 10km squares of the Broad Habitat species map (Figure 3.10). For nutrient nitrogen different empirical critical loads have been set for two EUNIS classes: dry heaths (F4.2) and Northern wet heaths (F4.11), the latter comprising *Calluna*-dominated and *Erica*-dominated wet heaths. Satellite imagery cannot identify individual species, nor separate areas of wet and dry heathland. The wet heath categories cannot be distinguished using species distribution data, since these are presence/absence data and both species tend to occur in the same grid squares. Therefore, a distinction has not been made between the two wet heathland types. The HOST data have been used to identify areas of wet and dry heaths which were then combined into a single "dwarf shrub heath" map (Figure 3.11) as only wet or dry heath can be mapped in any 1km grid square.

BAP Broad Habitat 10: Dwarf shrub heath

Critical load	EUNIS class	Habitat maps used
Acidity	F4 Temperate shrub heathland	Combination of LCM and species data
Nutrient nitrogen	F4.11 Northern wet heaths (*Calluna*-dominated upland wet heaths & *Erica*-dominated lowland wet heaths)	Combination of LCM and species data within areas of wet soils
	F4.2 Dry heaths	Combination of LCM and species data within areas of dry soils

Table 3.2 Division of the HOST classes into wet and dry soils

HOST class	Soil characteristics	Substrate hydrogeology	Groundwater or aquifer	Soil: Wet (W) Dry (D)
1	Mineral soil, no impermeable or gleyed layer within 100cm	Weakly consolidated, microporous, by-pass flow uncommon (chalk)	Normally present and at >2m	D
2	Mineral soil, no impermeable or gleyed layer within 100cm	Weakly consolidated, microporous, by-pass flow uncommon (limestone)	Normally present and at >2m	D
3	Mineral soil, no impermeable or gleyed layer within 100cm	Weakly consolidated, macroporous, by-pass flow uncommon	Normally present and at >2m	D
4	Mineral soil, no impermeable or gleyed layer within 100cm	Strongly consolidated, non or slightly porous, by-pass flow common	Normally present and at >2m	D
5	Mineral soil, no impermeable or gleyed layer within 100cm	Unconsolidated, macroporous, by-pass flow very uncommon	Normally present and at >2m	D
6	Mineral soil, no impermeable or gleyed layer within 100cm	Unconsolidated, microporous, by-pass flow common	Normally present and at >2m	D
7	Mineral soil, either no impermeable or gleyed layer within 100cm, or impermeable layer within 100cm or gleyed layer at 40-100cm	Unconsolidated, macroporous, by-pass flow very uncommon	Normally present and at <=2m	D
8	Mineral soil, either no impermeable or gleyed layer within 100cm, or impermeable layer within 100cm or gleyed layer at 40-100cm	Unconsolidated, microporous, by-pass flow common	Normally present and at <=2m	D
9	Mineral soil, gleyed layer within 40cm (IAC <12.5)	Unconsolidated, mircoporous, by-pass flow common	Normally present and at <=2m	W
10	Mineral soil, gleyed layer within 40cm (IAC >=12.5)	Unconsolidated, mircoporous, by-pass flow common	Normally present and at <=2m	W
11	Peat soil, drained	Unconsolidated, mircoporous, by-pass flow common	Normally present and at <=2m	D
12	Peat soil, undrained	Unconsolidated, mircoporous, by-pass flow common	Normally present and at <=2m	W
13	Mineral soil, impermeable layer within 100cm or gleyed layer at 40-100cm	Strongly consolidated, non or slightly porous, by-pass flow common	Normally present and at >2m	D
14	Mineral soil, gleyed layer within 40cm	Strongly consolidated, non or slightly porous, by-pass flow common	Normally present and at >2m	W
15	Peat soil	Strongly consolidated, non or slightly porous, by-pass flow common	Normally present and at >2m	W

HOST class	Soil characteristics	Substrate hydrogeology	Groundwater or aquifer	Soil: Wet (W) Dry (D)
16	Mineral soil, no impermeable or gleyed layer within 100cm	Slowly permeable	No significant groundwater or aquifer	D
17	Mineral soil, no impermeable or gleyed layer within 100cm	Impermeable (hard)	No significant groundwater or aquifer	D
18	Mineral soil, impermeable layer within 100cm or gleyed layer at 40-100cm (IAC >7.5)	Slowly impermeable	No significant groundwater or aquifer	W
19	Mineral soil, impermeable layer within 100cm or gleyed layer at 40-100cm (IAC >7.5)	Impermeable (hard)	No significant groundwater or aquifer	W
20	Mineral soil, impermeable layer within 100cm or gleyed layer at 40-100cm (IAC >7.5)	Impermeable (soft)	No significant groundwater or aquifer	W
21	Mineral soil, impermeable layer within 100cm or gleyed layer at 40-100cm (IAC <=7.5)	Slowly impermeable	No significant groundwater or aquifer	W
22	Mineral soil, impermeable layer within 100cm or gleyed layer at 40-100cm (IAC <=7.5)	Impermeable (hard)	No significant groundwater or aquifer	D
23	Mineral soil, impermeable layer within 100cm or gleyed layer at 40-100cm (IAC <=7.5)	Impermeable (soft)	No significant groundwater or aquifer	W
24	Mineral soil, gleyed layer within 40cm	Slowly impermeable	No significant groundwater or aquifer	W
25	Mineral soil, gleyed layer within 40cm	Impermeable (soft)	No significant groundwater or aquifer	W
26	Peat soil	Slowly permeable	No significant groundwater or aquifer	W
27	Peat soil	Impermeable (hard)	No significant groundwater or aquifer	W
28	Peat soil	Eroded peat	No significant groundwater or aquifer	W
29	Peat soil	Raw peat	No significant groundwater or aquifer	W

NB. HOST classes 18, 20, 21 and 23 may be dry soils in areas where agricultural drainage occurs. However, as the HOST data are being used to define habitats in non-agricultural areas, this should not pose a problem

3.7.4 Wetland habitats

The wetland BAP Broad Habitats considered for critical loads in the UK are (i) bogs, (ii) standing open water & canals and (iii) rivers and streams. Bogs are mapped for critical loads of acidity and nutrient nitrogen. Only a sub-set of UK standing open waters, rivers and streams are mapped (see below), and due to the nature of the sites selected they are considered in terms of acidification only.

BAP Broad Habitat 12: Bogs

Bogs were previously mapped as part of both the acid grassland and heathland ecosystems, by selecting those 1km squares dominated by peat soils. LCM 2000 has a specific class for bog habitats based on a combination of the satellite imagery and the British Geological Survey peat map. For this work, the habitat distribution has been further improved by overlaying the 10km species data for the bog Broad Habitat onto the LCM map, and then selecting the LCM areas within the 10km squares. The same map is used for mapping both acidity and nutrient nitrogen critical loads (Figure 3.12).

Critical load	EUNIS class	Habitat maps used
Acidity	D1 Raised and blanket bogs	Combination of LCM and species data
Nutrient nitrogen	D1 Raised and blanket bogs	Combination of LCM and species data

BAP Broad Habitat 13: Standing open water and canals
BAP Broad Habitat 14: Rivers and streams

Critical loads for freshwaters are based on the water chemistry samples for a large number of sites, consisting of a mixture of standing waters (lakes) and low-order streams, found largely in upland areas sensitive to acidification. Rigorous screening of the dataset used to map freshwater ecosystems has been undertaken in this update. The 'base set' consists of 1470 lakes and streams sampled by University College London under contract to the then Department of the Environment, together with additional data from a new critical loads survey in Northern Ireland, funded by DoE NI. The latter survey increased the number of sites included in critical loads mapping for Northern Ireland from 93 to 140, and replaced older chemistry data with results from the new survey undertaken in March 2000. Therefore, these critical loads data do not represent the whole of the two Broad Habitats (13 and 14), nor are they mapped separately, although the data have been submitted to the CCE for the separate EUNIS classes C1 (surface standing waters) and C2 (surface running waters).

The screening has resulted in the removal of a number of sites from the mapping dataset for a variety of reasons described below.

i) Charge balance error> ±10% or ±150$\mu eq l^{-1}$

Calculation of the charge balance (sum of positive ions minus sum of negative ions) was undertaken, making assumptions about the charge density of organic species measured as TOC. 67 sites for which there was a charge balance error greater than either ±10% or ±150$\mu eq l^{-1}$ were removed from the mapping dataset.

ii) Na:Cl ratio <0.4
Four sites with an apparent Na:Cl ratio of less than 0.4 were excluded as outliers in the dataset.

iii) Cl > 5000μeql^{-1}
Sites with chloride concentrations exceeding 5000 μeql^{-1} are defined as brackish and therefore inappropriate for freshwater critical loads modelling. 11 of these sites have been removed from the mapping dataset.

iv) Site specific factors
Rigorous inspection of site field sheets provided local information which justified the removal of a further 18 sites from the mapping dataset (e.g. observed presence of slurry drainage into sampled water bodies). Since the assumptions of the critical load models (e.g. marine origin of chloride, no direct inputs of N) do not hold if there are direct pollutant inputs to surface waters other than atmospheric sources, such sites have been excluded from further analysis.

v) Non-marine sulphate > 500μeql^{-1}
Critical loads models cannot be applied where there is evidence of non-marine sulphur inputs other than acid deposition. Since data are available on deposition input fluxes and estimated runoff fluxes of non-marine sulphate, it is possible to select a cut-off ratio of inputs to outputs to remove sites where non-atmospheric sources of S may be significant. This method assumes that the sulphate anion is mobile in terrestrial catchments but allows for uncertainty in calculation of input and output fluxes. A potential problem with this approach may occur at sites where stored sulphur is released into surface waters, for example by the oxidation of organic S. A conservative ratio of 2:1 (runoff to deposition flux for non-marine S) is therefore used as a pragmatic screening cut-off, i.e. where the leaching flux is more than double the deposition flux, sites have been removed from the mapping dataset because of potential non-atmospheric sources of S.

This final stage of screening reduces the number of mapped sites to 1044 for Great Britain and 119 for Northern Ireland. The total number of screened mapping sites for the UK is therefore 1163, compared with 1610 in the unscreened dataset and 1273 when only the non-marine sulphate cut-off concentration of 500 μeql^{-1} is used.

These specific freshwater sites are not considered sensitive to eutrophication and therefore only acidity critical loads are available for them.

Empirical nutrient nitrogen critical loads are defined in UNECE Mapping Manual for oligotrophic lakes and dune slack pools; however both of these habitats are too small to map at the national scale.

Critical load	EUNIS class	Habitat maps used
Acidity	C1 surface standing waters C2 surface running waters	Catchment boundaries of sites sampled under Freshwater Umbrella project
Nutrient nitrogen	C1.1 Oligotrophic lakes C1.16 Dune slack pools	Areas of these habitats too small to map at the national scale

3.7.5 Montane habitat

This BAP Broad Habitat includes moss and lichen dominated heaths of mountain summits, also represented by EUNIS class E4.2 (see below) for which empirical nitrogen critical loads have been set. However, this habitat cannot easily be mapped from satellite data alone. Additional information is required, such as species distributions and altitude. *Racomitrium* heath, found within montane habitats, is considered to be very sensitive to eutrophication and acidification. The 10km distribution map of the NVC class (U10) for *Carex bigelowii-Racomitrium lanuginosum* moss heath has been overlaid onto the LCM2000 data for the montane and inland bare ground classes. The LCM areas within the 10km squares have been selected and finally using a digital elevation model, any areas below 600m were excluded from the map (Figure 3.13).

BAP Broad Habitat 15: Montane

Critical load	EUNIS class	Habitat maps used
Acidity	E4.2 Moss and lichen dominated mountain summits	Combination of LCM data, NVC distribution map and altitude data
Nutrient nitrogen	E4.2 Moss and lichen dominated mountain summits	Combination of LCM data, NVC distribution map and altitude data

3.7.6 Coastal habitats

LCM2000 includes classes for the coastal BAP Broad Habitats littoral and supra-littoral rock and sediment. Whilst these habitats are not sensitive to acidification they can be sensitive to eutrophication and empirical nitrogen critical loads have been set for a number of coastal habitats (Chapter 7). For the UK only two of these habitats can be mapped, in combination, at the national scale. The LCM class areas for supra-littoral sediment (19.1) and littoral sediment (21.1) were added together. For this update, habitat experts have identified five key dune grassland species (*Ammophila arenaria, Carex arenaria, Corynephorus canescens, Leymus arenarius, Phleum arenarium*) and the distribution of these has been mapped for each 10km square across the UK. The LCM areas within any 10km square containing any of these species have been selected to represent the EUNIS classes for shifting coastal dunes and stable dune grassland (Figure 3.14).

BAP Broad Habitat 19: Supralittoral sediment

Critical load	EUNIS class	Habitat maps used
Acidity	None	None
Nutrient nitrogen	B1.3 Shifting coastal dunes B1.4 Coastal stable dune grassland	Combination of LCM data and distribution data of five key dune grassland species.

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4. CRITICAL LOADS OF ACIDITY FOR TERRESTRIAL HABITATS

4.1 Introduction

The National Expert Group on Transboundary Air Pollution (NEGTAP) collated evidence to show that acid deposition has resulted in acidification of acid sensitive soils in the UK. Although there is a paucity of long term monitoring data, there is now emerging evidence that the declines in acid deposition are leading to some recovery (e.g. Countryside Survey 2000, Haines-Young *et al.*, 2000). Recovery from acidification will be a slow process, as it is determined by base cation supply from atmospheric inputs and weathering.

Acidification is caused by nitrogen and sulphur. In the calculation of critical load exceedance maps, it is assumed that all nitrogen (derived from nitrogen oxides or ammonia) is acidifying in the long term. This is consistent with the critical load being a steady state concept (with long timescales being required to reach the steady state). However, there is still much debate within the scientific community to understand the fate of deposited nitrogen. The acidity critical load exceedance maps are considered a worst case scenario, and the future role of nitrogen deposition in acidification and recovery of soils remains an important research topic.

Two methods are used in the UK for calculating acidity critical loads for terrestrial habitats: the empirical approach is used to provide estimates for non-woodland habitats and a simple mass balance equation for woodland habitats. Both methods make use of the empirical critical loads of acidity for soils and this section begins with a description of the recent updates made to the latter.

4.2 Critical Loads of Acidity for Soils

Critical loads are assigned to each 1km square according to the dominant soil type occurring in each square. The critical loads are calculated using two methods: one for mineral and organic soils and another for peat soils. Both are described below. The combination of the critical loads for all soil types into a single map produces a map called the empirical critical loads of acidity for soils (Figure 4.1).

4.2.1 Empirical critical loads of acidity for non-peat soils

The methodology for calculating and mapping acidity critical loads for mineral and organic soils (Hornung *et al.*, 1995) remains unchanged from previous years. One of five critical load classes is assigned to each 1km grid square based on the mineralogy and weathering rate of the dominant soil (series or map unit) in each square. Each critical loads class is associated with a range of critical load values. However as a single value is usually required for each square, the mid-range value is used, with the exception with the critical loads in class 5 where the value is set to the top of the range (Table 4.1). This is consistent with work on soil weathering rates by Langan *et al.* (1995) and Sverdrup *et al.* (1990).

Table 4.1 Critical loads for non-peat soils

Critical loads class	Critical loads range (keq ha^{-1} $year^{-1}$)	Mid-range value used (keq ha^{-1} $year^{-1}$)
1	>2.0 <=4.0	4.0 (upper limit used)
2	>1.0 <=2.0	1.5
3	>0.5 <=1.0	0.75
4	>0.2 <=0.5	0.35
5	<=0.2	0.1

The soil surveys for England and Wales (National Soil Resources Institute - NSRI) and for Scotland (Macaulay Land Use Research Institute – MLURI) have revised their 1km resolution soil databases, which has led to some changes in the percentage areas of different soil types in each grid square. In addition, the revision of Scottish soils data by MLURI has led to a revision of those soils classified as peats. As the empirical map of soil critical loads is based on the dominant soil type, changes to the underlying soil databases have led to changes in the empirical map (see Chapter 8). No changes have been made to the 1km soils data for Northern Ireland, so the map for NI remains unchanged with the exception of the critical loads for peat-dominated squares (Section 4.2.2).

4.2.2 Acidity critical loads for peat soils

In this update, a change has been made to the calculation of acidity critical loads for peat soils. The updated method sets the critical load to the amount of acid deposition that would give rise to an effective rain pH of 4.4, whereas previously the critical load was based on the amount of acid deposition that would cause a fall of 0.2 pH units in the peat pH compared with the estimated pH of "pristine" peat. The rationale for the change in methods is given below.

Critical loads of acidity for peat soils are treated differently from those for mineral soils because of the absence of inputs of alkalinity from mineral weathering (Smith *et al.,* 1992; Gammack *et al.,* 1995; Hornung *et al.*, 1995a & 1995b). The calculation of acidity critical loads for peat soils is based on the concept of effective rain pH ie, total acidifying pollutant load divided by runoff.

Using regression equations relating peat pH to effective rain pH, critical loads were previously based upon the estimated amount of deposited acid that would cause a fall of 0.2 pH units in the peat pH compared with the estimated pH value of "pristine" peat. However, the selection of a value of 0.2 pH units for UK critical loads mapping was primarily chemistry-based, and was only very loosely related to biological impacts at the time. For example, work by Yesmin *et al.* (1995) suggests that this fall of 0.2 pH units may have been over-precautionary and that a substantial exceedance of the critical load would be necessary to induce a significant decline in the total enchytraeid population in *Calluna* moorland peats.

More recent research has demonstrated that the chemical and biological response to acidity in peat is more closely related to the threshold of 4.4 for effective rain pH: Yesmin *et al.* (1996) showed that the best correlation between transformed mycorrhizal infection of *Calluna* roots and deposition parameters was with effective rain pH; Dawod (1996), Proctor and Maltby (1998) and Parveen (2001) have shown that peat soil solution pH equals effective rain pH.

A review of the critical loads concept by Cresser (2000) concluded that for peat soils especially, critical load quantification could only sensibly be based upon the prediction of the pH of soil solutions. Such a method could then be meaningfully related to biological and physicochemical effects (Sanger *et al.*, 1996; Cresser *et al.*, 1997). Close scrutiny of the results of Proctor and Maltby (1998), as reproduced by Charman (2002) demonstrates that fitting a curve to their experimental data for peat pH versus effective rain pH is more appropriate than using linear regression, and results in an equilibrium value at ca. pH 4.4. This finding is borne out by Cresser and Calver (in preparation). This pH reflects the buffering effects of organic acids upon peat drainage water pH. There is no justification for attempting to protect the pH of peat soil solution to a value above this equilibrium threshold value. The evidence therefore suggests that critical loads of acidity for peat soils should be set at a value corresponding to the acid deposition load that would give rise to an effective rain pH value of 4.4. In this update, this has been applied in the calculation of acidity critical loads for all peat-dominated 1km squares in the UK using the equation:

$$CLA = q * [H^+]$$

Where:
q = runoff in metres
$[H^+]$ = critical hydrogen concentration equivalent to pH 4.4

The runoff data are the mean 1km values for 1941-1970; the same data set as used in the Simple Mass Balance equation for acidity critical loads for forest soils.

4.2.3 Other implications of the changes to the 1km soils data

The changes to the underlying 1km soils database has implications for other data sets used in the calculation of critical loads in this update:

- Acidity critical loads for non-woodland terrestrial habitats are based on the empirical soil acidity critical loads map (Section 4.3)
- Base cation weathering rates, used in the simple mass balance (SMB) equation for calculating acidity critical loads for woodland habitats (Section 4.4) are based on the empirical acidity critical loads map.
- Calcium weathering rates, also used in the SMB equation (Section 4.4.4) are also derived from the empirical acidity critical loads map.
- Identification of mineral versus organic soils (non-peat soils), for applying the correct criteria in the SMB equation (Section 4.4).
- Nitrogen immobilisation values used in the calculation of the minimum critical load of nitrogen (Chapter 5) and in the nitrogen mass balance equation (Chapter 7) are assigned according to the dominant soil type in each 1km grid square.
- Denitrification values used in the calculation of the minimum critical load of nitrogen (Chapter 5) and in the nitrogen mass balance equation (Chapter 7) are assigned according to the dominant soil type in each 1km grid square.

4.3 Empirical critical loads of acidity for non-woodland habitats

There has been no change to the method for non-woodland habitats, where the map of empirical critical loads of acidity for soils is used to set critical loads that will protect the soils on which the habitats depend. Changes to that empirical map (section 4.2) also apply here. In order to calculate parameters for the Critical Loads Function (Chapter 5), additional habitat-specific data are used.

4.4 Simple mass balance (SMB) acidity critical loads for woodland habitats

Critical loads for woodland habitats (Figure 4.2) are calculated using the simple mass balance (SMB) equation (summarised in Appendix 1A). In this update, the method remains unchanged, but more recent data have been used in the calculations. Different chemical criteria are used in calculating the critical loads for woodland on mineral soils and woodland on organic soils. The SMB calculation is applied to two BAP Broad Habitats: (i) Broadleaved mixed and yew woodland (comprising managed broadleaved woodland and unmanaged broadleaved and coniferous woodland); (ii) Coniferous woodland (comprising managed coniferous woodland only; the unmanaged component is included in (i)). Details on the woodland categories mapped are provided in Section 3.7.1. Critical loads are calculated for both managed and unmanaged woodlands in order to protect the long-term ecosystem function of the woodland habitats. Where woodland occurs in squares dominated by peat soils, the acidity critical loads for peats described above (Section 4.2.2), are used, as the SMB is considered inappropriate in such areas.

4.4.1 Chemical criteria

The choice of chemical criteria for critical loads for woodland soils remain unchanged since the last submission to the CCE (February 2001). Work by Hall *et al.* (2001a & 2001b) highlighted that the critical molar Ca:Al ratio in the soil solution is more appropriate for mineral soils than organic soils; for the latter a critical pH is considered to be more suitable. The UNECE Workshop on Chemical Criteria and Critical Limits (Hall *et al.*, 2001c) recommended critical soil solution pH as the preferred criterion for organic soils. The pH value of 4.0 is recommended in the UNECE Mapping Manual (UBA, 1996). Therefore, a critical molar soil solution Ca:Al ratio of one, is applied to forested mineral soils and a critical soil solution pH of 4.0 to forested organic (non-peat) soils.

4.4.2 Gibbsite equilibrium constant

The values for the gibbsite equilibrium constant (K_{gibb}), which simulate the relationship between aluminium and hydrogen ions in soil solution, remain unchanged since the last data submission (February 2001). The value applied to mineral soils is 950 m^6/eq^2 and the value for organic soils is 9.5 m^6/eq^2. These values are based on the percentage of organic matter in the soil and are recommended in the UNECE Mapping Manual (UBA, 1996).

4.4.3 Calcium deposition

The calculation of the acidity critical load for forested mineral soils, based on the Ca:Al criterion, requires total calcium deposition (wet plus dry, marine plus non-marine) values. More recent calcium deposition maps (for the year 2000) have been used[3].

4.4.4 Base cation and calcium weathering

Critical loads of acidity (CLA) are calculated in the SMB equation as:

$CLA = ANC_w - ANC_{le(crit)}$

where ANC_w = acid neutralising capacity (ANC) generated by base cation weathering

and $ANC_{le(crit)}$ = critical leaching of ANC

The empirical critical loads of acidity for soils (section 4.2), are based on the mineralogy and weathering rate characteristics of the dominant soil, and can therefore be used to provide ANC_w inputs to the SMB. The weathering rate is set to zero for those 1km grid squares dominated by peat soils.

The formulation of the SMB adopted in the UK for woodland on mineral soils uses a critical molar Ca:Al ratio of one in the soil solution as the chemical effects criteria. This means that the base cation terms in the calculation of $ANC_{le(crit)}$ need to be considered in terms of calcium only (Appendix 1A). As calcium weathering is a fraction of the total base cation weathering, estimates are obtained by applying "calcium correction" values to the base cation values:

Calcium weathering = ANC_w * calcium correction factor

The correction factors were provided by the NSRI and MLURI soil surveys, either by 1km grid square or by soil type. Some minor changes have been made to the calcium correction factors for England and Wales. Once again the weathering rate is set to zero for the peat-dominated squares.

4.4.5 Base cation, calcium and nitrogen uptake values

These uptake values are required for the calculations of critical loads for managed woodland habitats. For unmanaged woodlands all uptake terms are set to zero assuming that no harvesting, and therefore no removal of base cations, calcium or nitrogen takes place.

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[3] Calcium deposition is one of the measurements in the UK's Acid Deposition Monitoring Network, running since 1986. For 39 sites over the UK, rainwater samples are analysed for sulphate, nitrate, chloride, phosphate, sodium, magnesium, ammonium, calcium and potassium as well as pH and conductivity. A new ion chromatograph was installed in 2000. It has been found that the previous ion chromatograph underestimated calcium and magnesium concentrations. Although attempts have been made to correct data between 1993 and 1999, this has not proved successful. The recommendation is therefore that calcium values for 2000 should be used as the best estimate for all years between 1993 and 1999. Following further data from the new ion chromatograph, it may be possible to revise this recommendation in 2004.

The methods used to estimate base cation (Bc), calcium (Ca) and nitrogen (N) losses by uptake and removal during harvesting and thinning operations in forests and woodlands have been updated since those used for the February 2001 CCE data submission (Hall *et al.*, 2001d). New estimates of Bc, Ca and N uptake at harvest are based on site-specific measurements made at the ten UNECE/EU Intensive Forest Health monitoring sites (Level II) in the UK operated by Forest Research between 1995 and the present. The estimates of uptake are calculated using average volume increments (ie, a measurement of yield) which are converted into the amount (Bc etc) removed in harvest based upon the wood density and the concentrations in the wood. All calculations used the same equation:

Loss from site	=	average volume increment	*	basic wood density	*	concentration in wood
(keq ha^{-1} $year^{-1}$)		(m^3 ha^{-1} $year^{-1}$)		(g m^{-3})		(keq g^{-1})

Cumulative volume production including yield from thinnings are predicted from forest yield tables (Edwards and Christie, 1981). Rotation length is based on felling at maximum mean annual increment (MAI) for the two conifer species. In the case of oak, the rotation is extended beyond maximum MAI to 120 or 140 years to reflect typical practice. Overbark (ie, including bark) volumes (as given in the yield tables) are converted to underbark (ie, excluding bark) volumes using industry-accepted, species specific conversion factors (Hamilton, 1975) providing separate estimates of wood and bark volumes.

The three oak plots are assumed to be thinned, while of the conifer species, Sitka spruce is assumed unthinned, and Scots pine, thinned. The mean of the three broadleaved and seven conifer plots are then taken as representative values for their respective forest categories. The mean yield class of these two forest categories (5.0 m^3 ha^{-1} $year^{-1}$ broadleaf and 15.8 m^3 ha^{-1} yr^{-1} conifer) are higher than the average for the 0.9 MHa of the Forest Enterprise estate (3.2 and 11.6 m^3 ha^{-1} yr^{-1} respectively), and thus uptake values have been scaled accordingly.

Species specific densities for wood and bark (Lavers, 1969; Hamilton, 1975) are used to calculate biomass. For broadleaved species, branch biomass is calculated additionally, accounting for small diameter timber taken off site for pulp and firewood.

Site specific measured stemwood and bark nutrient concentrations together with published values of branch nutrient concentrations (Allen et al., 1974: for oak only) are then used to estimate total quantities of Ca, Bc and N taken offsite during the rotation. Uptake is assumed to occur at a constant rate over the course of the rotation.

In the base of Ca and Bc uptake by broadleaved species, two of the sites (Savernake and Alice Holt) are assumed to represent calcium-rich soils, and one (the Lakes), calcium poor soils. N uptake of broadleaved species was calculated as the mean of all three sites. Work is on-going with a view to further updating of the uptake values. This process will be augmented by data from an additional ten Level II sites, including beech and Norway spruce which have recently been established.

Table 4.1 Base cation, calcium and nitrogen uptake values for managed coniferous and broadleaved woodland

Woodland Type	Uptake Values (keq ha^{-1} $year^{-1}$) February 2001			Uptake Values (keq ha^{-1} $year^{-1}$) February 2003*		
	base cations	calcium	nitrogen	base cations	calcium	nitrogen
Conifers	0.25	0.12	0.5	0.27	0.16	0.21
Broadleaved Ca-rich soils	0.85	0.7	0.5	0.41	0.29	0.42
Broadleaved Ca-poor soils	0.4	0.33	0.5	0.315	0.195	0.42

* Conifer values based on the mean of four Sitka (Coalburn, Tummel, Loch Awe, Llyn Brianne) and three Scots pine (Thetford, Sherwood, Rannoch) sites. Broadleaved values for Ca-poor soils based on the Grizedale oak site and values for Ca-rich soils based on the mean of data for Alice Holt and Savernake oak sites

NB. Where the SMB is applied to unmanaged broadleaved and unmanaged coniferous woodland, all the uptake terms are set to zero, assuming that no harvesting takes place.

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5. ACIDITY CRITICAL LOADS FUNCTION (CLF) FOR TERRESTRIAL HABITATS

5.1 Introduction

Deposition of both sulphur and nitrogen compounds can contribute to exceedance of the acidity critical load. The Critical Load Function, developed under the UNECE CLRTAP (Posch *et al.*, 1999; Posch & Hettelingh, 1997; Posch *et al.*, 1995; Hettelingh *et al.*, 1995), defines combinations of sulphur and nitrogen deposition that will not cause harmful effects. The Critical Load Function (CLF) is - a three-node line on a graph representing the acidity critical load. Combinations of deposition above this line (the CLF) would exceed the critical load, while all areas below or on the line represent an "envelope of protection" where critical loads are not exceeded (Figure 5.1). Further information on the CLF is provided in Appendix 1.

The paragraphs below describe the methods used to calculate these critical loads and identifies where any changes have been made to the input data used. The calculation of these critical loads for freshwaters is dealt with separately in Appendix 1B.

5.2 Maximum Critical Load of Sulphur ($CL_{max}S$)

$CL_{max}S$ is based on the acidity critical load values but also takes into account the net base cation deposition to the soil system and base cation removal from the system:

$$CL_{max}S = CL(A) + BC_{dep} - BC_u$$

Where $CL(A)$ = acidity critical load (empirical or SMB)
$BC_{dep}{}^{*}$ = non-marine base cation less non-marine chloride deposition
BC_u = base cation uptake by vegetation

The national maps of $CL_{max}S$ for each broad habitat are shown in figures 5.2, 5.3 and 5.4. The acidity critical loads used in this calculation are those described in Chapter 4 Base cation deposition data for the year 2000 have been used (section 4.4.3).

The values of BC_u remain unchanged for all non-forest habitats. The value is set at zero for acid grassland, dwarf shrub heath , bog and montane., all based on Rawes & Heal (1978) and Reynolds *et al.* (1987). It is highlighted that previously dwarf shrub heath, bog and montane were not mapped individually. Dwarf shrub heath was previously included in the heathland ecosystem. Bog was included in both acid grassland and heathland, both of which have a BC_u value of zero. Montane areas were not specifically identified and are likely to have fallen within both the heathland and acid grassland ecosystems. The value for calcareous grassland remains unchanged at 0.222 keq ha^{-1} $year^{-1}$.

The distinction between managed and unmanaged woodland in this Update allows separate BC_u values to be assigned, whereas previously all woodland was assumed to be managed. For the managed woodland habitats the new BC_u values described in Section 4.4.5 and listed in Table 4.1, were used. For unmanaged woodland the uptake values were set to zero, assuming no harvesting, and therefore no base cation removal, is taking place.

5.3 Minimum Critical Load of Nitrogen ($CL_{min}N$)

$CL_{min}N$ is calculated as:

$$CL_{min}N = N_u + N_i + N_{de}$$

Where N_u = nitrogen uptake
N_i = nitrogen immobilisation
N_{de} = denitrification

The national maps of $CL_{min}N$ for each broad habitat are shown in figures 5.5, 5.6 and 5.7. In this update, changes to N_i and N_{de} arise from the changes in the map of empirical critical loads of acidity for soils, as they are based on the dominant soil type in each grid square. The values of N_i and N_d for each soil type have not changed: with N_i values of 1 and 3 kg N ha^{-1} year^{-1} and N_{de} values of 1, 2 and 4 kg N ha^{-1} year^{-1}, based on soil type.

In addition, N_i values have been amended for the dwarf shrub heath habitats to take account of nitrogen losses through fire (ie, N_{fire}), in accordance with the UNECE Mapping Manual (UBA, 1996). N_{fire} values of 4.5 kg N ha^{-1} year^{-1} in areas of wet dwarf shrub heath (EUNIS class F4.11 Northern wet heaths) and 15 kg N ha^{-1} year^{-1} in areas of dry dwarf shrub heath (EUNIS class F4.2 dry heaths) have been used.

These values have been derived from the available literature. For lowland heath in Dorset, Chapman (1967) calculates a total loss of 182.2 kg N ha^{-1} on burning a 12 year old stand. Dividing this by 12 gives an annual value of 15 kg N ha^{-1} year^{-1}. For blanket peat in the Pennines, 45 kg N ha^{-1} can be lost with a single burn (Allen. 1964), and the burn frequency for the Pennines varies between 7 and 20 years. Hornung (pers. comm. 2002) assumed an average burn frequency of 10 years, therefore deriving an annual value of 4.5 kg N ha^{-1} year^{-1} for upland heath. Based on this information it was agreed to use a value of 4.5 kg N ha^{-1} year^{-1} in areas of wet dwarf shrub heath and 15 kg N ha^{-1} year^{-1} in areas of dry dwarf shrub heath.

In this update, revisions have been made to the following nitrogen uptake values:

- The value for acid grassland has been changed from 1 kg N ha^{-1} year^{-1} to 1.14 kg N ha^{-1} year^{-1} based on data from Frissel (1978).
- The value previously used for heathland (4 kg N ha^{-1} year^{-1}) was considered to be too high. This view was supported by Perkins 1978, Rawes & Heal 1978, Reynolds *et al.* 1987, Batey 1982 and Gordon *et al.* 2001. From the literature a range of 0.5 to 1 kg N ha^{-1} year^{-1} was suggested, and a value of 0.5 kg N ha^{-1} year^{-1} was agreed as the value to be applied to the dwarf shrub heath habitat.
- For the bog habitat the same value as for dwarf shrub heath, has been used, ie, 0.5 kg N ha^{-1} year^{-1} (Reynolds, Woodin, pers. comm.).
- New uptake values for the managed woodland habitats are given in Section 4.4.5 and Table 4.1. For unmanaged woodland the uptake values were set to zero, assuming no harvesting, and therefore no nitrogen removal, is taking place.

The uptake value for calcareous grassland remains unchanged at 10 kg N ha^{-1} year^{-1}, though there remains some concern that this value is too high.

5.4 Maximum Critical Load of Nitrogen ($CL_{max}N$)

$CL_{max}N$ is calculated as:

$$CL_{max}N = CL_{min}N + CL_{max}S$$

Therefore, any changes to the input data used in the calculations of $CL_{min}N$ and $CL_{max}S$ can lead to changes in the values of $CL_{max}N$ (Chapter 8). Figures 5.8, 5.9 and 5.10 show the national maps of $CL_{max}N$ for each broad habitat.

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6. CRITICAL LOADS OF ACIDITY FOR FRESHWATERS

6.1 Introduction

Recent publications of the Acid Waters Monitoring Network (AWMN: Monteith & Evans, 2000) and NEGTAP (2001) have focussed on the response of natural systems to the reductions in sulphur deposition achieved over the last 15 years. Data from the AWMN have shown that although chemical change consistent with recovery is evident at many monitoring sites, only the very first stages of biological recovery have been detected, and only at a small number of these sites.

Current research efforts under the Freshwaters Umbrella programme (Defra contract, EPG 1/3/183) are focused on targets and timescales for recovery, in particular the processes responsible for the hysteresis in biological recovery. Updating of the national critical loads dataset for freshwaters also falls within the remit of the Freshwater Umbrella group, and new data on the effects of acid episodes, organic substances and dissolved aluminium are being used to refine the critical loads approach for freshwaters in the UK.

Although the science behind the critical loads models of acidity for fresh waters is well established (Appendix 1B), there is still debate over the most appropriate parameterisation of models developed in Scandinavia for UK conditions. One aspect of the debate centres on the choice of a single value of ANC_{crit} for diverse UK waters, while the other key aspect focuses largely on current understanding of nitrogen retention and leaching processes, especially the timescales over which nitrate may contribute to acidification or delay chemical and biological recovery as sulphate leaching declines.

6.2 Updates to parameters in FAB model

The FAB model (Appendix 1B) continues to be used for the calculation of critical loads of acidity for freshwaters (Figure 6.1), using the dataset of lakes and streams described in section 3.7.4. The dataset can be said to represent part, but not all, of two BAP Broad Habitats, with lake sites associated with "Standing open water and canals" and stream sites with "Rivers and streams". However, all sites are presented in a single UK map (Figure 6.1). Revisions to values for some of the parameters in the FAB model have been made in this update: forest uptake, denitrification, long term nitrogen immobilisation.

6.2.1 Forest uptake data

As described in section 3.7.1, the maps of woodland cover have been updated, and now make the distinction between managed and unmanaged. The freshwater critical load data set therefore incorporates new N_{upt} data based on new woodland cover data. Only managed woodland, where there is net removal of biomass from the catchment, provides a long-term sink for N. Also, data on the coverage of managed broadleaved woodland is included for the first time. Previous FAB model applications have used coniferous woodland coverage only, and assumed that all of it is managed and providing a potential sink for N. A net N uptake value of 5.88 kgN ha^{-1} yr^{-1} is assumed for managed broadleaved woodland. The default value for net N uptake in managed coniferous forest has also been updated to 2.94 kgN ha^{-1} yr^{-1} (see section 4.4.5)

6.2.2 Denitrification data

The UK FAB model application employs default values for denitrification (Hall *et al.*, 1998, see Appendix 1B). Recent work under Defra's Freshwater Umbrella (Curtis, 2001; Curtis, 2003), suggests that the default values are much more appropriate than the UNECE Mapping Manual (UBA, 1996) suggested method of 10-80% denitrification, determined by percentage peat cover. The assumption of very high denitrification rates in peat soils disguises the fact that most retained N in mass balance models is probably immobilised in soils rather than denitrified.

6.2.3 In lake retention component

It has been suggested by critics of the FAB model that the default mass-transfer coefficients employed in FAB, derived from studies in large Canadian lakes, may be inappropriate for UK conditions, and could underestimate net in-lake retention. However, there is no evidence from existing data to suggest that in-lake retention of S and N is being under-estimated in the model, and if anything it may actually be over-estimated. Monthly measurements of water chemistry in upland lake inflows and outflows over 2 years under a previous DoE contract (CLAG) indicated very little difference in concentrations of nitrate or sulphate from the inflow to the outflow, except for a time lag related to lake retention time. Norwegian studies (Berge *et al.*, 1997) have also found negligible in-lake retention in acid-sensitive upland lakes. It may be true that in-lake retention can be significant in eutrophic lowland lakes but there is little to suggest this is a major sink for acidity in oligotrophic, acid-sensitive upland lakes in the UK. It is true that FAB ignores denitrification in rivers, but no major rivers are included in the UK freshwaters mapping dataset - sites are either standing waters or low order streams. Denitrification from streams is not quantified but there is no evidence that this is a major sink for N in the low-order, acid-sensitive upland streams which show critical load exceedance.

6.2.4 Long-term N immobilisation

Results from the first Freshwaters Umbrella contract (Curtis, 2001; Curtis 2003) suggest that current rates of N_{imm} are much higher than the long-term default values provided in the UNECE Mapping Manual (UBA, 1996). This phenomenon is well known (and stated in the Mapping Manual) and presents a key problem for parameterisation of mass balance models for N. The major uncertainty is related to the process of N saturation and the capacity of a catchment to assimilate N through time until increased N leaching occurs, i.e. what is the sustainable rate of N immobilisation under enhanced N deposition? This question is particularly difficult to address because of the complex dynamics of N, links to the carbon cycle, and the potentially long timescales involved, superimposed on a situation of ecological and climatic change. The process is not yet sufficiently well understood to allow adaptation of steady-state models, so default values continue to be used (see Hall *et al.*, 1998).

6.3 Choice of ANC_{crit} (0, 20 μeql^{-1} or variable ANC?)

No change has been made to the value of critical chemical criterion (ANC = 0 μeql^{-1}) in this update. However, questions remain over the most appropriate choice of ANC_{crit} for UK surface waters, because of their great variety in terms of water chemistry and catchment hydrology. The currently used value of 0 μeql^{-1} is deemed too low for many sensitive sites to facilitate rapid biological recovery, but the wholesale adoption of a more precautionary value of 20 μeql^{-1} has been hampered by uncertainties about the possible presence of lakes with natural ANC values below this threshold, for which critical loads could never be achieved.

Various options are being explored for the application of variable ANC_{crit} methods according to different criteria, including threshold values for labile aluminium, alkalinity and TOC. Such methods are, however, subject to problems of chemical change through acidification which affects the critical load, removing the time invariant characteristic required of static critical loads. Other options, such as the use of catchment soils data to classify relative risk of acid episodes with high concentrations of labile aluminium, have also been explored and are currently ruled out because of limitations with soils data resolution.

Work is currently underway to evaluate and determine the effects of new ANC_{crit} values on exceedances in the UK national mapping freshwater dataset. Despite continuing work on the biological response of freshwaters to acidification in the UK, it appears that there are still insufficient data to adapt the Norwegian dose-response curve for brown trout (Lien *et al.*, 1996) to UK conditions. Hence it is still not possible to determine whether ANC_{crit} of 0 μeql^{-1} does indeed provide a 50% probability of damage to brown trout populations for UK waters. Although recent publications have shown that streams are likely to require a higher mean ANC_{crit} to provide the same level of protection (i.e. likelihood of damage) as that given to lakes because of the much greater impacts of acid episodes in streams, there is not the required spatial and temporal coverage of chemical and biological data to quantify the required ANC_{crit}. Hence there is currently no strong case on a purely scientific basis for recommending a change to an ANC_{crit} value of 20 μeql^{-1} for all UK freshwaters. Therefore, at present, an ANC_{crit} of 0 μeql^{-1} will continue to be used.

However, the Freshwater Umbrella group has voiced its concern over the maintenance of the status quo for ANC_{crit} in the light of increasing concerns over impacted species other than brown trout and the lack of clear signs of recovery in many biological groups in extensive areas of the acid uplands (e.g. Wales). A key issue is the question of whether a 50% probability of damage to brown trout is acceptable for surface waters in areas of conservation or recreational interest. Furthermore, organisms other than brown trout are of great importance in many surface waters within Special Areas of Conservation (SACs) and some of these are potentially more sensitive to acidification than brown trout, as shown by previous work reviewed under an ongoing Environment Agency project (EA R&D Project Ref 12094: Freshwater Screening and Assessment Based on Freshwater Critical Loads). For example, salmon have long been known to be more acid sensitive than brown trout and the biodiversity in macrophyte communities of naturally acidic upland lakes may be reduced even where acidification has not reduced ANC to zero.

As more research findings are published on the response of freshwater biota to acidification and acid episodes, there are two questions which should be considered.

1. Should a higher ANC_{crit} be selected to protect organisms that may be more acid sensitive than brown trout?
2. Is a 50% probability of damage an acceptable level, given increasing concerns in areas of conservation interest in the context of the EU Habitats Directive?

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7. CRITICAL LOADS OF NUTRIENT NITROGEN FOR TERRESTRIAL ECOSYSTEMS

7.1 Introduction

7.1.1. Ecological and Ecosystem Responses to Nitrogen Deposition

Nitrogen is the main soil derived nutrient and plays a major role in plant and ecological processes. Through industrial and agricultural activity, humans have significantly increased the conversion of inert N_2 into reactive chemical forms of nitrogen. These compounds may all be assimilated by plants and soils and contribute to their nitrogen demand. As increasing amounts of pollutant nitrogen become available plants and soils suffer from an excessive supply or 'eutrophication'. The optimum amount of nitrogen required varies widely for different systems and this gives rise to the range of proposed critical loads for habitats within the UK. While agricultural crops are unlikely to be directly affected by typical rates of nitrogen deposition, many natural and semi-natural ecological communities are more sensitive because nitrogen is the main limiting nutrient. These systems, such as heaths, moors, bogs and grassland, are adapted to low nutrient supply and the plants survive and compete successfully in these impoverished conditions.

The ultimate consequence of an excessive nitrogen supply to nutrient-poor communities is a shift in the composition of the community so that nitrogen-sensitive plants are lost and an overall reduction is seen in biodiversity. The mechanisms through which nitrogen causes these changes are many, owing to the different N pollutant forms deposited, the contrasting plant receptors and the diverse range of processes in which nitrogen is involved. The potential effects of nitrogen are discussed in NEGTAP (2001), and are summarised below.

- Direct toxic effects of nitrogen pollutants on above ground parts of plants resulting in poor growth and performance

- Accumulation of nitrogen compounds in soil and subsequent increase in their availability to plants causing change in plant community composition

- Increased susceptibility of plants to secondary stress and disturbance factors such as frost, drought, pathogens and herbivores.

- Increased leaching of nitrogen from soils into waters with consequences for stream water chemistry and aquatic biota

- Acidification of soils leading to nutrient imbalance and changes in plant community composition

7.1.2 Critical loads for nutrient nitrogen

This wide range of possible impacts means that different types of critical load may be appropriate for use, depending on the impacts of concern. UNECE recommend two main approaches to calculating critical loads for nutrient nitrogen. The first is a steady state mass balance approach (section 7.5) in which the long-term inputs and outputs of nitrogen from the system are calculated, with the critical load being exceeded when any excess nitrogen input is calculated to lead to exceedance of a critical rate of nitrogen leaching. The steady state mass balance for nutrient nitrogen is calculated as:

$$CL_{nut}N = N_u + N_i + N_{le(acc)} + N_{de}$$

Where N_u = nitrogen uptake

N_i = nitrogen immobilisation

$N_{le(acc)}$ = acceptable level of nitrogen leaching

N_{de} = denitrification

This mass balance approach is applied to managed woodlands (section 7.5) to ensure the long-term ecosystem function (eg, soils, soil biological resources, trees and linked aquatic ecosystems) is protected.

The second is an empirical approach, in which critical loads are estimated, rather than calculated, for different ecosystems based on experimental or field evidence of thresholds for change in species composition, plant vitality or soil processes. The mass balance approach is better suited to managed ecosystems of low biodiversity, in which inputs and outputs can be quantified with some confidence and in which the key concern is nitrate leaching. The empirical approach is better suited to semi-natural communities for which the long-term protection of biodiversity and/or ecosystem function is the key concern. For these reasons, the UK chooses to use both mass balance and empirical approaches in mapping critical loads across the country, applying them to different types of ecosystem. Since the empirical critical loads are applied to a wider range of communities, these are considered first.

7.2 Empirical Critical Loads for Nutrient Nitrogen

7.2.1. Introduction

This update of UK mapping values for empirical critical loads of nitrogen has been closely linked to a review of the empirical critical loads recommended for application across Europe. This review resulted in the adoption of new recommended empirical critical values at a formal UNECE workshop held in Berne in November 2002. In order to provide a basis for discussion of the revised values of critical loads, a comprehensive review of new literature was commissioned by UNECE, from which proposals were made for revised critical load values (Bobbink *et al.*, in press). This review was presented at the Berne meeting and provided the basis for discussion by experts at the meeting, and for the final recommendations for empirical critical loads to be applied within UNECE. A revised version of the review of Bobbink et al. (in press) is currently in draft, taking into account the results of discussions at the Berne workshop. UK experts provided input to, and scientific review of the Bobbink *et al.* (in press) document, and actively participated in the Berne workshop. There is high confidence, therefore, that this international review process has taken full account of latest UK research and the views of UK experts.

Because of the rigour of the review process underlying the new values for empirical critical loads recommended at the Berne workshop, and the active engagement of UK experts in the process, these critical load values have been used in revising recommended empirical critical loads for national mapping in the UK. In terms of UNECE, the values recommended at the Berne workshop replace those which were adopted in 1996, based on the earlier review of Bobbink *et al.* (1996). The need for the formal revision of these values was based primarily on a preliminary assessment of new scientific evidence at a UNECE workshop held in York in March 2000 (Hall *et al.*, 2001). In 2000, the National Expert Group on Transboundary Air Pollution (NEGTAP) recommended some changes to the empirical critical loads for UK mapping, based on the results of the York workshop and its review of eutrophication in the UK (NEGTAP, 2001).

This part of the UK Status report aims to summarise:-

- the outcomes of the Berne workshop
- how these outcomes have been used to recommend new mapping values for empirical critical loads of nitrogen in the UK
- where appropriate, how the adoption at the Berne workshop of the EUNIS classification (section 3.4) in defining communities with differing critical loads has altered the UK mapping process for critical loads.

This section of the report aims to summarise the process by which the revised critical loads for the UK were defined. More details on the impacts of nitrogen deposition, the principles through which empirical critical loads are defined, and the evidence underlying the recommendation of specific critical loads at the Berne workshop can be found in NEGTAP (2001) and Bobbink *et al.* (in press). An important component of the evidence underlying the values of empirical critical loads are field manipulation experiments, in which plots are exposed to different rates of simulated nitrogen deposition under field conditions. An updated review of such field manipulation experiments, with an emphasis on UK studies, was recently published by the Joint Nature Conservation Committee (Cunha *et al.*, 2002). This review provides further details of specific experiments that are mentioned in Section 7.3.

Table 7.1 summarises, for non-forest ecosystems, the recommendations from the Berne workshop, alongside the critical loads which were used for European mapping prior to the Berne workshop. For forest ecosystems, a rather different approach was adopted, as described in Section 7.4. In some cases, the values in Table 7.1 remain unchanged, in some cases revised values have been adopted, and in other cases new critical loads have been identified for new communities. Empirical critical loads are not recommended by UNECE for all plant communities – ***the focus is on those communities which are likely to be sensitive to nitrogen deposition and which have a distribution which makes them significant nationally and internationally. This therefore may exclude communities which are of local high conservation value***.

It is important to note in interpreting Table 7.1 that:-

1. The critical loads are expressed as a range rather than a single value. This range indicates the real variation in sensitivity within a particular ecosystem, for example, because of differences in nutrient status, management etc.

2. That the uncertainty in the critical load range is expressed qualitatively, by assessing the critical load as being reliable, quite reliable and expert judgement. It is important to note that this represents a judgement of the extent and quality of the scientific evidence available from which critical loads might be estimated.

Even where the evidence is classed as reliable, there may be different views on its interpretation and therefore on the appropriate critical load range.

3. The background document provides some guidelines on how local site conditions might influence the appropriate value of the critical load. The relevance of this guidance to specific UK communities is explained in more detail in Section 7.3.

The empirical critical loads recommended by UNECE for each community are defined as a range, rather than a single value. This means that mapping values within the range for each community need to be defined for the UK. In order to do this, a set of basic principles have been used which are described and discussed in the next section.

7.2.2. *Principles underlying the choice of UK mapping values for empirical nutrient nitrogen critical loads*

In theory, it would be possible to use information about the environmental factors influencing the critical loads within the range of the UNECE Berne recommendations, alongside appropriate national datasets, to provide national maps with a range of critical load values for each ecosystem. However, the current judgement of UK experts is that, while qualitative statements can be made about the influence of modifying factors such as phosphorus status, management and climate on the critical load, there is not an adequate basis on which to make quantitative estimates of the effects of these factors across the country. Therefore a single mapping value has been selected for use in mapping critical loads across the UK. However, these modifying factors might be taken into account when evaluating the appropriate critical load for a specific site, for which more detailed information may be available.

In deciding which value to use for UK mapping, it is important to consider whether it is possible to map the national distribution of a specific target ecosystem. A key change in the recommended values adopted at the Berne workshop was the use of the EUNIS classification, which allows a standard European approach to ecosystem classification. The adoption of this new method of ecosystem classification, and the fact that new ecosystems have been identified for inclusion, means that significant changes have to be made to UK mapping procedures. These issues are considered in more detail in terms of defining critical loads in Section 7.3; those issues related to mapping EUNIS categories were considered in Section 3.5.

The proposed mapping values for non-forest ecosystems in the UK are summarised in Table 7.1, alongside the recommended ranges of critical loads adopted at the Berne workshop, and the mapping values previously used for the UK (NEGTAP, 2001). The reasons for the choice of the new mapping values are explained in more detail in the individual sections for each type of ecosystem in Section 7.3. However, at these stage, some general points of principle can be identified:-

- For those critical loads identified as expert judgement, a mapping value is not recommended unless there is a specific body of evidence of relevance to the UK, and it refers to a significant UK plant community. However, in some cases the recommended values are low, and there is clearly an urgent need for research to test their validity.

- UK mapping values which are not in the middle of the UN/ECE range are recommended where the field or experimental evidence from the UK specifically suggests that this is not appropriate

- Values other than the mid-range have in some cases been recommended where knowledge of UK ecosystems suggests that they are more or less sensitive than the median for this ecosystem across Europe.

- When there is no specific UK evidence to suggest otherwise, the middle of the range from the Berne workshop is recommended for UK mapping.

Table 7.1: SUMMARY OF 2000 AND REVISED EMPIRICAL CRITICAL LOADS FOR NITROGEN (kg ha^{-1} yr^{-1}) FOR NON-FOREST ECOSYSTEMS

(a) Ecosystem (with corresponding EUNIS class, where used)	(b) 2001 UK mapping value	(c) Critical load range in 1996 Mapping Manual	(d) Critical load range from Berne workshop	(e) Revised UK mapping value
Grasslands				
Dry acid and neutral closed grassland (E1.7)	25	20-30 #	10-20 #	15
Calcareous grassland (E1.26)	25[(1)]	15-35 #	15-25 ##	20
Montane grassland	12	10-15 (#)		
Hay meadows (E2.2)			20-30 (#)	-
Montane hay meadows (E2.3)			10-20 (#)	-
Arctic/sub-alpine grass (E4.3 & E4.4)			10-15 (#)	-
Moist/wet oligotrophic grass (E3.5)			10-20 #	15
Molinia meadows (E3.51)			15-25 {#)	-
Nardus stricta swards (E3.52)			10-20 #	15
Moss/lichen mountain summits (E4.2)			5-10 #	7
Inland dune pioneer grass (E1.94)			10-20 (#)	-
Inland dune silicaceous grass (E1.95)			10-20 (#)	-
Heathland/moorland				
Lowland dry heaths (F4.2)	17	15-20 ##	10-20 ##	12
Lowland *Erica* wet heaths (F4.11)		17-22 #	10-25 #	15
Upland *Calluna* wet heaths (F4.11)	15	10-20 (#)	10-20 (#)	15
Arctic/alpine heaths (F2)	7.5	5-15 (#)	5-15 (#)	-
Tundra (F1)			5-10 #	-
Coastal habitats				
Coastal stable dune grasslands (B1.4)		20-30 #	10-20 #	15
Shifting coastal dunes (B1.3)			10-20 #	15
Coastal dune heaths (B1.5)			10-20 (#)	-
Moist-wet dune slacks (B1.8)			10-25 (#)	-
Dune slack pools (C1.16)			10-20 (#)	-
Salt marshes (A2.64 & A2.65)			30-40 (#)	-
Softwater oligotophic lakes				
Permanent oligotrophic lakes (C1.1)		5-10 ##	5-10 ##	-
Bogs, mires and fens				
Ombrotrophic and raised bogs (D1)	10	5-10 #	5-10 ##	10
Poor fens (D2.2)			10-20 #	15
Rich fens (D4.1)			15-35 (#)	-
Montane rich fens (D4.2)			15-25 (#)	-

The reliability of the recommended range is indicated as:-
(#): expert judgement
#: quite reliable
##: reliable

The tabulated values are taken from the following sources:-
(b) NEGTAP (2001); (c) Bobbink *et al.* (1996); (d) Bobbink *et al.* (in press); (e). This document

[(1)] 25 kg N ha^{-1} $year^{-1}$ was the value recommended by NEGTAP (2001). However, this decision was taken after the data submission in February 2001, in which a value of 50 kg N ha^{-1} $year^{-1}$ was applied.

7.2.3. Interpretation of Exceedance of Empirical Critical Loads of Nitrogen

Exceedance of critical loads simply indicates that there exists a potential for adverse effects of nitrogen pollution on the most sensitive elements within particular habitats. Currently, it is not possible to state unequivocally either the timescale, or to quantify the extent, of the change that is likely to occur in the majority of habitats. These constraints reflect the highly complex functions of nitrogen in habitats and ecosystems. For example, while we can quickly recognise replacement of nitrogen sensitive species by more competitive nitrophilic ones, it is very difficult to predict, and indeed to detect, longer term impacts of such changes in individual species composition on wider community functions such as nutrient or carbon cycling and plant-animal interactions.

Indicators of adverse change resulting from nitrogen deposition vary from responses of individual plants, through to alterations in soil and water chemistry and increased leaching from ecosystems. The key indicators of exceedance used to define empirical critical loads at the Berne workshop, and as discussed in NEGTAP (2001), are principally changes in plant community composition, since these are often the prime conservation interest. However, research into higher plant-soil systems indicates that the soil microbial community is at least equally sensitive to nitrogen supply, and that changes in microbial activity that influence nutrient availability may both pre-empt the initial plant response and also have a longer lasting influence on key soil processes and ecosystem function. Evidence of changes in soil biological activity are therefore valuable indicators of ecological change, underpinning, and in some cases explaining, observations of plant response above ground. In addition, there is a need to protect the whole ecosystem, including the soil, from nitrogen enrichment in order to sustain future ecosystem health and function.

The benefits of reductions in nitrogen deposition to UK ecosystems are likely to be apparent over lengthy time scales. Evidence from experiments and observations in the field indicate that much of the nitrogen deposited to terrestrial systems in recent decades has accumulated in soil compartments and it is likely to be retained there for a long time, even if future deposition is dramatically lowered. The build-up of nitrogen in the soil potentially stores up problems for N-limited communities and will increasingly push the balance in favour of nitrogen demanding competitor species. Future environmental conditions resulting from climate change may also accelerate the mobilisation of stored nitrogen resulting in adverse changes in soil chemistry and increased leaching.

In the light of long-term nitrogen accumulation within ecosystems, what are likely to be the benefits of reduced emissions? In the UK, the decline in NOx emissions since about 1990 has not, so far, yielded any obvious improvements in ecological health of sensitive habitats (NEGTAP, 2001). However, this is probably not surprising, since NH_3 releases have changed little over that time and the modest changes in total nitrogen emissions must be seen against a background of rising inputs over the past two centuries or so.

It is likely that the first signs of improvement in terrestrial ecosystems will be seen in the performance, or distribution, of mosses, lichens and annual flowering plants as these should be the most sensitive to a lower deposition or aerial concentration of anthropogenic nitrogen. Evidence for recovery in mosses has been seen in an experiment at CEH Bangor using mesocosms of grass communities exposed to lower than ambient nitrogen deposition (NEGTAP, 2001). In the Netherlands, some benefits of falling emissions of nitrogen compounds, particularly of ammonia, since the early 1990s have been observed in terms of floristic changes in nitrogen-sensitive habitats (Roelofs, pers. com.). In some habitats, recovery can be accelerated by removal of the organic soil nitrogen pool achieved through turf stripping, a common practise already on some lowland heaths in the UK, which is used extensively in the Netherlands. However, such operations are clearly less practical in other habitats, such as bogs and many upland systems. Other management tools, such as controlled burning, mowing and grazing potentially offer alternative means of removing the stored nitrogen pool. However, recent research suggests that they will only be effective if they reduce nitrogen levels in the soil compartment, where the majority of the additional nitrogen is stored (Caporn *et al.*, 2002; Power *et al.*, 2001).

Active research in the UK and elsewhere is currently aiming to detect and predict the ecological impacts of future reductions in nitrogen emissions. In advance of obtaining clear signs of improvement in the UK as a consequence of reductions in nitrogen emissions, we should at least be confident that a lowering of nitrogen deposition will help guard against further deterioration in the quality of ecosystems and habitats. This is also an important consideration for the many areas of UK semi-natural vegetation where the critical load remains exceeded.

7.2.4 Field evidence of impacts of deposited N

The purpose of this document is to explain how revised critical loads of nutrient nitrogen have been developed. However, it is of relevance to consider the extent to which evidence exists of impacts of nitrogen deposition on sensitive communities in the UK. Deposition rates for nitrogen in many areas of the UK exceed the critical loads summarised in Table 7.1, indicating the potential for long-term impacts of nitrogen deposition across the country (NEGTAP, 2001). Evidence of impacts of nitrogen deposition in the field provides important support for the significance of such exceedance of critical loads. However, it is important to emphasise that the lack of such evidence cannot invalidate the critical loads because:-

(i) the study design may not be adequate to detect effects of N deposition;
(ii) the long-term nature of responses to deposited nitrogen means that adverse effects may occur at some point in the future; and
(iii) local modifying factors may reduce the impacts of N deposition at a specific location.

Three types of field evidence exist:-

1. Evidence of changes in species composition, growth or vitality through time. Key issues in the interpretation of such evidence are the continuity in location of the plots, the measurement methods, and the role of other factors such as site management in causing the observed change.

2. Evidence of spatial associations between nitrogen deposition and species composition and other responses. A key issue in the interpretation of such evidence will be the confounding effects of factors such as climate. The strongest

evidence of cause-effect relationships from spatial associations will be close to point sources of pollution. For example, Pitcairn *et al.* (1998) reported a gradient study of ground flora composition in an acid woodland away from an intensive livestock unit and found a greater frequency of nitrophilic species above an estimated deposition rate of 15-20 kg ha^{-1} yr^{-1}.

3. Evidence that the nitrogen content of foliage has increased over time in areas with high levels of nitrogen deposition. There is evidence of increases in the nitrogen content of mosses and heather in many areas of the UK over the last few decades, which is consistent with a cumulative effect of nitrogen deposition (e.g. Pitcairn *et al.*, 1995).

The evidence of changes in foliar nitrogen content and of species composition in the UK, and their links to experimental evidence, were critically reviewed and summarised by NEGTAP (2001). The clearest evidence of a national signal of the impacts of nitrogen deposition on species composition across the UK countryside comes from the Countryside Vegetation Survey (CVS) (Haines-Young *et al.*, 2000). These data have been analysed using the Ellenberg fertility index which rates each species in terms of whether it has a high or low demand for nutrients such as nitrogen. Analysis of the CVS survey data for the period 1990-98 showed a significant increase in the Ellenberg index of ground flora in some upland, but not lowland, woodland plots, and in infertile grassland, moorland grass, and heaths and bogs. This suggests a decrease in the cover of species adapted to low nitrogen conditions. As expected, these changes were not observed in fertile grasslands or arable plots, which were already dominated by species with a high nutrient demand.

A recent more detailed analysis of these data has assessed whether the change in Ellenberg fertility index showed any significant relationships with nitrogen deposition, i.e. whether there had been greater reduction in the cover of species adapted to low nitrogen conditions in areas with high nitrogen deposition. The results indicated that this was the case for grasslands and for heaths and bogs, but not for woodlands (Smart *et al.*, pers. comm.). Furthermore, analysis of individual species data for heaths and bogs over the period 1978-90 showed that the probability of a decrease in cover for the three dominant ericaceous species: *Calluna vulgaris*, *Erica tetralix* and *Erica cinerea* was associated with modelled deposition of reduced nitrogen. Experiments and field studies have shown that these nitrogen-sensitive species are liable to be replaced by grass at high rates of nitrogen deposition.

While this, and other, field evidence is not conclusive proof of the national impact of nitrogen deposition on UK communities, it is consistent with the fact that critical loads for these more sensitive communities are exceeded over significant areas of the UK. The field evidence is also broadly consistent with the changes observed in shorter-term field manipulation experiments, which are important in providing support for empirical critical loads (NEGTAP, 2001; Cunha *et al.*, 2002).

7.3 UK Mapping values for non-forest ecosystems

A key conclusion of the Berne workshop was the adoption of different approaches to assigning critical loads for forest and non-forest ecosystems. For non-forest ecosystems, different critical loads are assigned to different ecosystem types. In contrast, the emphasis for forests was to identify critical loads for different responses which might be applied to a range of forest communities. In this section, the main non-forest classifications (grasslands, heathlands, wetlands and coastal habitats) are considered in turn. It is important to note that the list of ecosystems in Table 7.1 is limited, and does not cover all EUNIS categories. This is because:-

1. For some major ecosystem types, primarily in Mediterranean and eastern Europe, there is simply no information from which to estimate a critical load value.
2. The focus is on ecosystems which are likely to be affected by enhanced nitrogen deposition – hence many ecosystems of high nutrient status are omitted
3. Many ecosystem types are too localised for national and international mapping, and therefore, despite a low nutrient status, have no critical load assigned.

Those ecosystem types under (3) will include some of high conservation value, which require protection under the Habitats Directive. The implications of this for the conservation agencies and for local site assessment are beyond the scope of this document, which is concerned with national mapping of critical loads.

7.3.1. Grasslands (Figure 7.1)

Bobbink *et al.* (in press) note the overwhelming range of grassland types across Europe. The Berne workshop recommended empirical critical loads only for a small number of these grassland types, identified as being of low nutrient status, for which some evidence was available. Furthermore several of the critical load ranges for grasslands that were recommended at the Berne workshop were either based only expert judgement, were for communities not relevant to the UK, or were too localised for national mapping. Therefore, in proposing new mapping values for the UK only four categories were considered. These fall within the broad EUNIS categories of dry grasslands (E1), wet grasslands (E3) and alpine/subalpine grasslands (E4); no critical load is recommended for UK mapping for mesic grasslands (E2). These mesic grasslands include the improved grasslands which dominate many areas of the lowland UK landscape. While more detail on each category is provided below, it can be pointed out at this stage that, for all four categories, no strong argument was identified to vary from the policy of using the middle of the critical load range agreed at the Berne workshop.

Sub-Atlantic semi-dry calcareous grassland (E1.26)

A wide range of critical load values for calcareous grassland (15-35 kg ha^{-1} yr^{-1}) was recommended by Bobbink *et al.* (1996). A basic assumption underlying this range was that the impact of N deposition in calcareous grasslands was through an increase in plant growth, which led to increased competition, and loss of sensitive species adapted to lower nutrient levels. On this basis, the higher end of this range referred to those grasslands (including most of those in the UK) where phosphorus supply was limited. On such grasslands, it was assumed that there would be no response of plant growth to additional N deposition, and hence no increase in competition and no loss of species. The intention was that the high end of the range should be used in P limited systems and the lower end of the range when P was not limiting.

However, more recent experimental data detailed in Cunha *et al.* (2002), including work in the UK, has shown that significant effects on vegetation composition (increases in grasses and decreases in herbs, legumes and geophytes) can occur on P-limited grasslands in response to increased N deposition. The cover of typical bryophyte species has also declined in experimental studies in response to increased N deposition. These changes occur even though total plant growth and vegetation cover is decreased, rather than increased, in response to increased N supply. This new evidence suggests that direct toxicity, rather than competitive exclusion as originally hypothesised by Bobbink *et al.* (1996), may be responsible for the observed responses. It is also important to note that these experimental studies, and associated model simulations, suggest that these changes may take 5-15 years to become apparent, even at relatively high deposition rates.

The new evidence of the response of P-limited calcareous grasslands led to a reduction in the critical load recommended at Berne to 15-25 kg ha^{-1} yr^{-1}, removing the higher end of the previous range. Given that some of the evidence supporting this revised critical load is based on experimental studies in P-limited UK grasslands (see Cunha *et al.*, 2002), it is appropriate to apply the mid-range value of the range recommended at the Berne workshop, i.e. 20 kg ha^{-1} yr^{-1} (see Figure 7.1a).

Non-Mediterranean dry acid and neutral closed grassland (E1.7)

In the previous process of assigning critical loads adopted by Bobbink *et al.* (1996), dry acid grasslands with high species diversity were assigned a critical load of 10-15 kg ha^{-1} yr^{-1}, whereas other acid and neutral grasslands were assigned a higher critical load of 20-30 kg ha^{-1} yr^{-1}. The adoption of the EUNIS system has meant that the categorisation of grassland communities has changed, and this factor needed to be recognised alongside the new experimental and field evidence, at the Berne workshop. In particular, dry grasslands (E1) are distinguished from mesic grasslands (E2). The latter are likely to have a higher critical load, corresponding to the value of 20-30 kg ha^{-1} yr^{-1} for other grasslands proposed by Bobbink *et al.* (1996), but this is only based on expert judgement. Therefore, such mesic grasslands, although an extensive component of UK vegetation, do not have a mapping value assigned to them. ***It is also important to note that all the evidence considered by Bobbink et al. (in press) in recommending a critical load for EUNIS class 1.7 is derived from acid, rather than neutral, grassland communities.***

New mesocosm studies, including very low rates of nitrogen deposition, suggest that significant changes in species composition occur above 20 kg ha^{-1} yr^{-1} after two years. Furthermore, new evidence from a long-term field experiment in the UK shows that significant effects on species composition of higher plants occur at all applications above current deposition in the southern Pennines (about 20 kg ha^{-1} yr^{-1}) after ten years, and in species composition of mosses after two years (Bobbink *et al.* (in press), Cunha *et al.* (2002)). On the basis of this, and other, evidence a critical load range of 10-20 kg ha^{-1} yr^{-1} was adopted at the Berne workshop. The middle of this range (15 kg ha^{-1} yr^{-1}) is recommended as the UK mapping value (Figure 7.1b). Since the evidence supporting this critical load was based on acid grassland studies, only acid grassland communities are mapped when assessing exceedance of this critical load; dry neutral grassland communities are not mapped.

Moist and wet oligotrophic grasslands (E3.5), including Nardus stricta *swards (E3.52)*
Recent field and mesocosm studies on upland oligotrophic communities in the UK provide a basis for setting a specific critical load for these communities; this was not done by Bobbink *et al.* (1996), because of the lack of relevant data. In both field and mesocosm studies, significant changes in higher plant species composition, along with increased nitrate leaching, were found above 20 kg ha^{-1} yr^{-1} after 3-4 years of treatment. In contrast, effects on the cover and species composition of the moss component were found below 20 kg ha^{-1} yr^{-1}, and in some cases below 10 kg ha^{-1} yr^{-1}. The results from these experiments were used at the Berne workshop to recommend a critical load of 10-20 kg ha^{-1} yr^{-1}. Since UK experiments provided the main empirical evidence for this range, it is logical to use the middle of the range (15 kg ha^{-1} yr^{-1}) for UK mapping (see Figure 7.1b).

Moss and lichen dominated mountain summits (E4.2)
Within the EUNIS system, E4.2 is a sub-category of arctic and sub-alpine grassland without extensive snow cover, which is dominated by moss and lichen communities. The only evidence available to Bobbink *et al.* (in press) to support the recommendation of a critical load for this EUNIS category was UK evidence related to *Racomitrium* heath, which is a community of high conservation value found on mountain summits in Britain. There is evidence in the UK of a serious decline in these communities over the past two decades, which may at least partly be associated with increased nitrogen deposition, as well as changes in grazing pressure and other factors (Pearce & van der Val, 2002).

Field experiments in the Scottish Highlands (Pearce & van der Val, 2002; Cunha *et al.*, 2002) have shown large declines in *Racomitrium* growth and cover, with increased grass cover, at N addition rates within two years at only 10 kg ha^{-1} yr^{-1} above the estimated current deposition rate of 12 kg ha^{-1} yr^{-1}. The fact that these were very large and rapid effects, which are consistent with observed field declines, led to a precautionary approach in assessing the empirical critical load at the Berne workshop. Based on this study, and other supportive evidence, the critical load range adopted at the Berne workshop was 5-10 kg ha^{-1} yr^{-1}. Given that the evidence supporting this critical load was from UK studies, a value in the middle of the range (7 kg ha^{-1} yr^{-1}) has been adopted for UK mapping (Figure 7.1c).

7.3.2. Heathland/moorland
Critical loads for UK mapping in this category refer to temperate heathland (EUNIS class F4). The recommendations from the Berne working group considering heathlands and moorlands identified a number of modifying factors of specific relevance to these systems. These were:-

1. Management. These semi-natural systems are essentially maintained by some form of management. The intensity of management can lead to the removal of different amounts of nitrogen from the system, but can also affect responses to nitrogen in more subtle ways.

2. Phosphorus availability. In general, when soil availability of phosphorus is limiting, a smaller degree of response to additional atmospheric nitrogen is expected.

When applying these critical loads to specific sites, it is clear that the nutrient status of the site and its current management need to be considered in order to identify the appropriate value within the range. Indeed changes in management practice may effectively alter the sensitivity of the site to atmospheric deposition and hence change the local critical load. For national mapping purposes, we cannot take this site-specific approach. Therefore, the following reasoning has been used.

Dry heaths (F4.2)

The previous value of 15-20 kg ha^{-1} yr^{-1} (Bobbink *et al.*, 1996) for dry heathland was based on long-term model simulations based on Dutch data, which assumed management by sod-cutting every 50 years. These models, which describe competition between one of two ericaceous shrubs, and one of two invasive grass species, were based on extensive experimental and field research and hence are reliable. Discussion at the Berne workshop focussed on whether this narrow critical load range was appropriate for application to all European dry heathland.

Research in the UK, in which these models have been parameterised for UK dry heaths, has clearly demonstrated that the critical load for conversion of *Calluna*-dominated wet heaths into acid grassland communities is reduced under conventional UK management, such as mowing, by 5 kg ha^{-1} or more, compared with sod-cutting. In addition, an experiment of over ten year's duration, on a UK lowland heath (see Cunha *et al.*, 2002), shows significant effects on growth, nutrient budgets and lichen flora, of the addition of N deposition at 8 kg ha^{-1} yr^{-1}. UK maps suggest that N deposition at this site is about 15 kg ha^{-1} yr^{-1}. However, recent site–specific measurements by Power & Barker (2003) report a value of 8 kg ha^{-1} yr^{-1}, excluding aerosol and nitric acid deposition

Based on this new evidence from modelling and from experimental studies, the Berne workshop proposed a reduction in the lower end of the critical load range, giving a range of 10-20 kg ha^{-1} yr^{-1}. Given that the higher end of this range is based on model simulations using intensive sod-cutting management, not widely practised in the UK, and taking account of the field experiment in the UK which suggests a threshold of 8-15 kg ha^{-1} yr^{-1}, a UK mapping value of 12 kg ha^{-1} yr^{-1} (towards the lower end of the new range) is recommended (Figure 7.2).

Northern wet heath (F4.11)

This EUNIS class can be divided into *Calluna* dominated wet heath, characteristic of the uplands, and *Erica tetralix* dominated wet heath, more characteristic of lowland heaths. For both categories, expert judgement was indicated as the basis of the critical load agreed at the Berne workshop. However, given the high cover of this community in the UK, and the existence of long-term field experiments in this country, a mapping value is required for the UK.

For *Calluna*-dominated wet heath, or moorland, a long-term field manipulation experiment in North Wales shows significant effects of adding 40 kg ha^{-1} yr^{-1} to a background deposition of 25 kg ha^{-1} yr^{-1} over a period of ten years. Long-term effects at lower rates of deposition have not been examined, and therefore it is difficult to identify an effect threshold for this community. The recommendation of experts at the Berne workshop was that the same critical load range (10-20 kg ha^{-1} yr^{-1}) should be set for *Calluna* dominated wet and dry heaths. This decision is supported by comparisons of experimental studies on wet and dry heaths in the UK, in which the nature of the observed responses were comparable (Cunha *et al.*, 2002). The modifying effect of management in wet heath communities is less certain than for dry heath, as, in terms of grazing in particular, the interactions may be complex. Therefore, there is no basis for recommending the use in the UK of either end of the critical load range and a mapping value is recommended of 15 kg ha^{-1} yr^{-1}, the middle of the range of 10-20 kg ha^{-1} yr^{-1} adopted at the Berne workshop.

For *Erica tetralix* dominated heath, the range recommended at the Bern workshop was 10-25 kg ha^{-1} yr^{-1}, based on the results of modelling studies conducted in the Netherlands. However, the upper end of this range was based on simulation models using the intensive management of sod-cutting and therefore is inappropriate for the UK. A mapping value of 15 kg ha^{-1} yr^{-1} is therefore recommended for the UK (Figure 7.2).

7.3.3. Wetlands

EUNIS class D covers a wide range of wetland communities. However, only for two broad classes – raised and blanket bogs and poor fens – are the critical loads based on experimental and field evidence, rather than expert judgement.

Raised and blanket bogs (D1)

The assessment for the workshop at Berne provided significant new empirical evidence to support the previous critical load range of 5-10 kg ha^{-1} yr^{-1}. At the lower end of the range, the key response expected is a change in species composition within the bryophyte layer. In contrast, at the higher end of the range it was considered that the capacity of the bryophyte layer to absorb deposited nitrogen would be saturated, and increased nitrogen availability in the rooting zone will lead to increased competition from vascular plants.

A key modifying factor for bogs considered at the Berne workshop was precipitation. In general terms, evidence suggests that the impact of N deposition would be lower in areas where precipitation is high. This is important, as much of the key evidence used to support the critical load in terms of changes in bryophyte species composition comes from regions of Europe, especially the Netherlands and Scandinavia, where precipitation is much lower than in bog areas of the UK. In contrast, the evidence for N saturation of the bryophyte layer comes from data at an international range of sites with different levels of precipitation (Bobbink *et al.*, in press). It is concluded that the upper end of the proposed critical load range is applicable to the UK, but that the lower end may not be appropriate at our high precipitation rates. Hence, a UK mapping value of 10 kg ha^{-1} yr^{-1} is recommended (Figure 7.3).

Phosphorus status was also identified at the Bern workshop as an important factor modifying responses of bog systems, but this will depend on the response of interest. Differential growth responses between bryophyte species, and hence changes in species composition, may be less likely in P-limited sites, but the capacity of the bryophyte layer to absorb deposited nitrogen may conversely be lower at such sites. Therefore, no specific use has been made of phosphorus availability in recommending UK mapping values.

Valley mires, poor fens and transition mires (D2)

A new critical load of 10-20 kg ha^{-1} yr^{-1} was proposed for poor fens and related communities at the Berne workshop, based on new field experiments in Scandinavia and northern continental Europe. There is no reason to expect differences in response between these communities and poor fens in the UK, and hence a mapping value of 15 kg ha^{-1} yr^{-1} could be applied. However, these communities are too localised in the UK for national mapping of critical load exceedance to be possible.

7.3.4. Coastal Habitats

In previous exercises, no UK mapping of coastal habitats has taken place. However, new evidence of effects on a range of dune communities, including evidence from the UK which was important in setting the revised critical loads for dunes at the Berne workshop, has led to a revision of this view. These studies suggest that changes in species composition may occur in coastal stable dune grasslands (EUNIS class B1.4) at rates of N deposition above 15 kg ha^{-1} yr^{-1}, leading to a recommended critical load range of 10-20 kg ha^{-1} yr^{-1}. Responses at similar deposition rates are expected in shifting coastal dunes. These were the only coastal communities for which the critical load ranges adopted at Berne were based on experimental and field evidence rather than expert judgement, and therefore these are the only coastal communities for which a UK mapping value is recommended. The UK mapping value (15 kg ha^{-1} yr^{-1}) has been set at the middle of the range (Figure 7.4). This decision is supported by recent field surveys in Wales and SW England, which suggest that several chemical and biological indicators can be exceeded at deposition rates above the proposed mapping value (Jones *et al.*, 2002).

7.4. Summary of new empirical critical loads for nitrogen for forest ecosystems

Up to the Berne workshop in November 2002, critical load values recommended for forests had differentiated between broad woodland types (e.g. deciduous/coniferous, acid/calcareous) and major types of chemical and biological response within forests (Bobbink *et al.*, 1996). This approach was abandoned at Berne, because it was considered to be impossible to differentiate the threshold for responses in different types of woodland. This also meant that, unlike for non-forest systems, the EUNIS system is not applied in differentiating empirical critical loads of nitrogen. Instead, it was proposed to define critical load ranges for different types of chemical and biological response (Bobbink *et al.*, in press). The Berne workshop also suggested a single critical load value for forests of 10-20 kg ha^{-1} yr^{-1}, as quite reliable, and suggested that national experts used broad guidance on modifying factors to decide on national mapping values. However, an approach based on specific response variables was considered to be more appropriate for use in the UK.

Critical load ranges were recommended for seven types of response at the Berne workshop. Four of these are not adopted for UK mapping. These relate to effects on

(i) mycorrhizae and (ii) pathogen susceptibility, which are both based on expert judgement, and (iii) nutrient imbalances (iv) and nitrification and mineralisation, as these cannot be related directly to responses in community composition or ecosystem function. It is therefore proposed to use effects on nitrate leaching, ground flora and epiphytic lichens and algae, for defining critical loads in the UK.

The critical load ranges adopted at Berne for these three responses are summarised in Table 7.2, with an indication of the woodland types within the UK for which these responses might be most relevant, and the revised UK mapping values. Table 7.2 also summarises the critical load ranges from Bobbink *et al.* (1996) and the 2000 UK mapping values, where these can be compared.

In deciding the UK mapping values for forests, it is important to remember that the steady-state mass balance approach is also available for application. This approach appears to be a more appropriate basis for assessing effects on nitrate leaching, but cannot be applied to effects on ground flora or epiphytes. This is consistent with the previous approach within the UK. The method and parameterisation of the steady-state mass balance approach in UK mapping is described in Section 7.5. A value of 12 kg ha^{-1} yr^{-1}, in the middle of the recommended range for nitrate leaching from the Berne workshop, could be applied at specific sites where the data to make steady-state mass balance calculations are not available.

In the case of the response of ground flora, field studies reviewed by Bobbink *et al.* (in press) supporting the critical load of 10-15 kg ha^{-1} yr^{-1} were located in temperate forests of southern Sweden and northern continental Europe. A single value was adopted, rather than the differentiated values with wider ranges proposed by Bobbink *et al.* (1996), as there is little clear evidence that deciduous and coniferous forests differ in terms of the threshold for effects on ground flora. The mid-range value of 12 kg ha^{-1} yr^{-1} is proposed for UK mapping (Figure 7.5a), in the absence of any specific evidence that UK forests differ in terms of response to N from continental forests. Specific field evidence from the UK to support this critical load is lacking, although effects on ground flora have been observed above 20 kg ha^{-1} $year^{-1}$ close to point sources of ammonia. Further field evidence is therefore urgently needed.

Prior to this update (ie, in February 2001), UK maps were produced by applying the steady-state mass balance approach (SSMB) to all UK woodlands, and mapping the minimum of this value and the empirical critical load of 12 kg ha^{-1} yr^{-1} for ground flora. This resulted in maps based on a combination of ecosystem effects.

However, an approach which is more consistent with the difference between these chemical and biological responses is that they are applied to the most relevant woodland types. In general terms, the steady-state mass balance is best applied to managed woodlands, with regular harvesting, where ground flora diversity is low, and is dominated by the management cycle. The ground flora empirical critical load, likewise in general terms, is best applied to mature woodlands with no regular felling and with high ground flora diversity. However, mapping the distribution of these woodland types is problematic, as was discussed in Chapter 3, and in some cases different management regimes may be practised in close proximity within the same wood.

The background document for the Berne workshop identified a number of field studies in Scandinavia which suggested that changes in epiphytic species composition could be detected at deposition rates in the range 5-10 kg ha^{-1} yr^{-1} a value which is consistent with the expert judgement made by Bobbink *et al.* (1996) (cf. Table 7.2). However, each of these field studies is subject to problems of interpretation because the gradients of nitrogen deposition are confounded by climatic gradients or gradients in deposition of other pollutants, and concerns were raised that the deposition rates in these studies may have been under-estimated. Therefore, the Berne workshop recommended a value of 10-15 kg ha^{-1} yr^{-1} as expert judgement, to prevent loss of sensitive epiphytic lichen species and to prevent increased cover of epiphytic algae.

Although this value is only expert judgement, and uncertainty over interpretation of these and other field studies remains, the epiphytic lichen and bryophyte flora of the UK is of very high international significance. It is therefore recommended that a mapping value of 10 kg ha^{-1} yr^{-1}, at the lower end of the range proposed at Berne, is used in the UK (Figure 7.5b). Application of this value to woodlands with a high diversity of epiphytes (eg, Atlantic oak woods) would provide added protection above the ground flora mapping value of 12 kg ha^{-1} yr^{-1}.

7.5 Steady state mass balance critical load for nutrient nitrogen

As explained above, the steady-state mass balance (SSMB) method is used to provide critical loads to protect the long-term ecosystem function of the woodland habitats. In this update the SSMB continues to be used, but is now only applied to managed (coniferous and broadleaved) woodlands. The critical load is calculated as:

$$CL_{nut}N = N_u + N_i + N_{le(acc)} + N_{de}$$

Where N_u = nitrogen uptake
N_i = nitrogen immobilisation
$N_{le(acc)}$ = acceptable level of nitrogen leaching
N_{de} = denitrification

The national maps of $CL_{nut}N$ for managed coniferous and broadleaved woodland are shown in Figure 7.6. The data and values used for N_u are given in Table 4.1 (Section 4.4.5), and N_i and N_{de} are described in Section 5.3. Changes have been made to the value of $N_{le(acc)}$ (previously a value of 6 kg N ha^{-1} $year^{-1}$ was used for both coniferous and deciduous woodland). In this update, for managed conifers, a range of 1-5 kg N ha^{-1} $year^{-1}$ was considered, with a single value of 4 kg N ha^{-1} $year^{-1}$ used in the calculation of $CL_{nut}N$, supported by Emmett *et al.* (1993) and Emmett & Reynolds (1996). For managed broadleaved woodland, a range of 1-3 kg N ha^{-1} $year^{-1}$ (Emmett, submitted) was considered, and a value of 3 kg N ha^{-1} $year^{-1}$ has been used in the calculation of $CL_{nut}N$.

Table 7.2: SUMMARY OF PROPOSED NEW EMPIRICAL CRITICAL LOADS FOR NITROGEN (kg ha^{-1} yr^{-1}) FOR FORESTS

Response and forest type	**(b) 2001 UK mapping value**	**(c) Critical load range in 1996 mapping manual**	**(d) Critical load range recommended at Berne workshop**	**(e) Updated UK mapping value (Feb 2003)**	**Updated UK woodland receptors (Feb 2003)**
Nitrate leaching Coniferous Deciduous	SSMB	-	10-15 ## 10-15 (#)	SSMB	Managed woodland ecosystems
Ground flora Coniferous Deciduous	13 17	7-20 ## 10-20 #	10-15 # (all forests)	12 (all forests)	Unmanaged woodlands with high diversity of ground flora
Epiphytic lichens and algae	-	5-10 (#)	10-15 (#)	10	Woodlands with high diversity of epiphytic lichens

Notes:

SSMB: steady state mass balance

The reliability of the recommended range is indicated as:-
(#): expert judgement
#: quite reliable
##: reliable

The tabulated values are taken from the following sources:-
(b) NEGTAP (2001); (c) Bobbink *et al.* (1996); (d) Bobbink *et al.* (in press); (e). This document

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8. Overview of changes to UK critical load maps

8.1 Introduction

Chapters 3 to 7 of this Report document the updates made to the data and methods used to calculate and map critical load values for acidity and nutrient nitrogen. This chapter presents an overview of the resultant critical load maps, and summarises the main changes.

The main areas that have been updated in the critical loads maps are:

i) choice of habitats for which critical loads are mapped
ii) maps of distribution of habitats
iii) changes to underlying data
iv) changes in methods to calculate or assign critical loads

This chapter aims to describe and illustrate the main changes. A selection of maps is presented here. It would be possible to produce a very large number of maps showing different combinations of the various changes. However, it should be noted that the change in the habitat classes for which critical loads are assigned (ie, from ecosystem classes to BAP broad habitats) means a like for like comparison between the previous and updated critical loads maps is not always straightforward.

Maps of the previous (Feb 2001) and updated (Feb 2003) critical loads, mapped according for each habitat type, are presented in Appendix 4 (see below). The main areas of change are described in Sections 8.2-8.5 and the implications of these changes on critical load values are described in Section 8.6 and 8.7.

8.2 Choice of habitats for which critical loads are mapped

For the February 2001 and earlier data submissions to the CCE, critical loads were mapped for 6 ecosystems considered sensitive to acidification and/or eutrophication. For this update, critical loads are mapped for BAP broad habitats sensitive to acidification and/or eutrophication, where sufficient data are available to enable the habitats to be mapped at the national scale. The changes to the choice of habitats are summarised in Table 8.1 below. The differences in the previous and updated mapping methods (ecosystems vs habitats) are summarised in Section 8.3.

Table 8.1 Ecosystems and Habitats mapped for critical loads

Feb 2003 (Updated) Broad habitat		**Feb 2001 (Previous) Ecosystem**	
Acidity broad habitat	***Categories within broad habitat mapped for nutrient nitrogen***	***Acidity ecosystem***	***Categories within ecosystems mapped for nutrient nitrogen***
Acid grassland	wet acid grassland dry acid grassland	Acid grassland	sub-alpine grass bogs
Calcareous grassland	as for acidity	Calcareous grassland	as for acidity
Dwarf shrub heath	wet heathland dry heathland	Heathland	upland *Calluna* moor arctic/alpine heath bogs
Standing open water, rivers & streams	not mapped	Freshwaters	not mapped
Coniferous woodland (managed areas only)[1]	as for acidity	Coniferous woodland (no distinction between managed/unmanaged)	as for acidity
Broadleaved woodland (managed areas only)	as for acidity	Deciduous woodland (no distinction between managed/unmanaged)	as for acidity
Broadleaved, mixed & yew woodland (unmanaged areas only)	Atlantic oak woods separated from other unmanaged areas[2]	Not mapped	Not mapped
Montane	as for acidity	Not mapped	Not mapped
Bog	as for acidity	Not mapped	Not mapped
not mapped	Supralittoral sediment	Not mapped	Not mapped

[1]Unmanaged coniferous woodland (including native Scots Pine) is included in the "Broadleaved, mixed & yew woodland (unmanaged)" habitat (see Section 3.7.1).
[2]Empirical nitrogen critical loads applied separately to Atlantic oak woods (to protect epiphytic lichens) and to the ground flora in all other unmanaged woodland.

8.3 Habitat distribution maps

The methods used to map the BAP broad habitats for acidity and nutrient nitrogen critical loads are described in detail in Chapter 3. The following paragraphs explain the differences in the methods used for mapping ecosystems in 2001 and broad habitats in 2003. It should also be noted that the previous maps (Feb 2001, mapped by ecosystem) excluded 1km grid squares where the ecosystem occupied <5% of the area (Hall *et al.*, 1998 & 2001), whereas the new broad habitat maps include all 1km squares where the habitat is present.

8.3.1 Acid grassland ecosystem vs Acid grassland broad habitat *(Figure 8.1)*
LCM1990 did not have a land cover class for acid grassland, so an aggregation of eight land cover classes were used to map the area referred to as "acid grassland ecosystem" for the earlier data submissions. However, some of the classes included in the map, would now be associated with heathland on LCM2000. The total area of acid grassland in the UK mapped in this way was 55784 km^2.

The acid grassland broad habitat map, produced in this update, is based on the LCM2000 class for acid grassland, further refined by species data for this broad habitat. The total area mapped for acidity is 15621 km^2 and for nutrient nitrogen 15241 km^2; the latter is slightly smaller because the HOST data used to distinguish wet and dry acid grassland do not have values for all the 1km habitat squares mapped

(Section 3.7.2). Both of these are significantly smaller than the previous map (above), because (a) they focused on a more specific habitat(s), and (b) they no longer include land cover classes associated with other habitats.

8.3.2 Calcareous grassland ecosystem vs Calcareous grassland broad habitat (Figures 8.2 & 8.3)

As for acid grassland, LCM1990 did not have a specific class for calcareous grassland. Instead classes for pasture/amenity grass and meadow/verge/unimproved grassland were selected and the area refined using key species distribution data for calcareous grassland. Areas where the empirical acidity critical load for soils were ≤ 0.5 keq ha^{-1} $year^{-1}$ were excluded from the map, on the basis that calcareous grassland should not occur in these areas (see Section 3.7.2). The total area of the calcareous grassland ecosystem is 24978 km^2; the area is very high because of the high percentages associated with the grassland land cover classes used.

For the broad habitat of calcareous grassland in this update, the specific LCM2000 class for this habitat is used. Once again, the area is refined using species data, but this time the species data include all species associated with the broad habitat. This map (Figure 8.2) is used for mapping nutrient nitrogen critical loads for calcareous grassland. To map this habitat for acidity critical loads a further refinement is made, excluding areas where the empirical acidity critical load for soils were ≤2.0 keq ha^{-1} $year^{-1}$ (Figure 8.3). This is done on the basis that calcareous grassland is not likely to occur on the acid soil dominating the 1km square and upon which the empirical acidity critical load is based (see Section 3.7.2). The total areas of calcareous grassland broad habitat are 1812 km^2 for acidity (Figure 8.3) and 3577 km^2 for nutrient nitrogen (Figure 8.2). The values are significantly lower than for the calcareous grassland ecosystem, because there are more grassland classes on LCM2000, and this refinement has meant that in this update, a single class is used.

***8.3.3 Heathland ecosystem vs Dwarf shrub heath broad habitat** (Figure 8.4)*

The heathland ecosystem map was based on an aggregation of four LCM1990 classes and had a total area of 10211 km^2. Some of the LCM1990 classes that would now be considered as heathland, were previously mapped as part of the acid grassland ecosystem, hence this area is smaller than the updated map (below).

The broad habitat map, in this update, is based on two LCM2000 classes and further refined with species data for this habitat. With the improved distinction between grassland and heathland classes on LCM2000, the area of the dwarf shrub heath broad habitat at 25748 km^2 is considerably larger than the previous heathland ecosystem map. The area mapped for nutrient nitrogen critical loads is slightly lower (24820 km^2) because the HOST data used to distinguish between wet and dry heath do not contain values for all 1km habitat squares mapped.

***8.3.4 Coniferous woodland ecosystem vs coniferous woodland broad habitat** (Figure 8.5)*

The coniferous woodland ecosystem map was based on a single LCM1990 class for coniferous woodland. The total area in the UK was 7464 km^2; this included all woodland, both managed and unmanaged.

The broad habitat map, in this update, is for managed (productive) coniferous woodland only; unmanaged conifers (including native Scots Pine) are included in the unmanaged component of the "Broadleaved, mixed and yew woodland" broad habitat (see Section 3.7.1). For GB managed coniferous woodland is based on the co-occurrence of the LCM2000 coniferous woodland class and Forest Research woodland inventory data for managed woodland. The ratio of FR managed to FR total woodland is applied to the LCM2000 data to estimate the managed woodland area for the purposes of critical loads. For NI, data are not available to distinguish between managed and unmanaged woodland, so the LCM2000 values are used, assuming all coniferous woodland in NI are managed (see Section 37.1). The total area is 8165 km^2 for acidity and 7979 km^2 for nutrient nitrogen; the area is slightly smaller for the latter because the nitrogen immobilisation and denitrification data used in the nitrogen mass balance equation do not have values for all 1km habitat squares mapped.

8.3.5 Deciduous woodland ecosystem vs broadleaved, mixed and yew woodland broad habitat (Figure 8.6)

The deciduous woodland ecosystem map was based on two LCM1990 classes (deciduous woodland and scrub/orchard), giving a total area of 10511 km^2. No distinction was made between managed and unmanaged woodland.

The broadleaved, mixed and yew woodland broad habitat, in this update, is sub-divided into two categories for mapping acidity critical loads:

- managed (productive) broadleaved woodland
- unmanaged (ancient and semi-natural) coniferous and broadleaved woodland including Atlantic oak woods

and into three categories for mapping nutrient nitrogen critical loads:

- managed (productive) broadleaved woodland
- unmanaged (ancient and semi-natural) coniferous and broadleaved woodland,
- Atlantic oak woods.

Refer to Chapter 3 for more details. For the managed and unmanaged woodland habitats, the maps have been based on the co-occurrence of LCM2000 and FR woodland inventory data. The Atlantic oak woods are a sub-set of the unmanaged woodland and were defined using a combination of LCM2000, FR woodland inventory and National Vegetation Classification data (see Section 3.7.1). The areas are: managed broadleaved woodland 7617 km^2, unmanaged coniferous and broadleaved woodland (including Atlantic oaks) 4209 km^2, giving a total of 11826 km^2.

8.4 Changes to underlying data

Changes made to the data underpinning the UK critical load calculations are documented in detail elsewhere in the report. Table 8.2 summarises these changes.

8.5 Changes in methods to calculate or assign critical loads

Only two changes have been made to the methods to calculate or assign critical loads and these are documented in detail elsewhere in the report. They are:

- Calculation of acidity critical loads for peat soils
- Assignment of empirical critical loads of nutrient nitrogen

8.6 Changes in acidity critical load values

Revisions to the underlying data sets and methods (see above) lead to changes in the mapped critical loads. The effect of changing the underlying 1km soils database for GB on the map of acidity critical loads for non-peat soils can be seen in figure 8.7. Changes are most apparent in East Anglia, south Wales and central England. The new method for calculating acidity critical loads for peat soils has increased the mean critical load value from 0.1 keq ha^{-1} $year^{-1}$ to 0.226 keq ha^{-1} $year^{-1}$ (Figure 8.8). It is this change in the peat critical loads that has the largest effect on the map of empirical acidity critical loads for (all) soils in the UK (Figure 8.9). However, overall the mean critical load value on this map (Figure 8.9) has only increased by 0.04 keq ha^{-1} $year^{-1}$.

Changes in the input values to the SMB acidity critical load maps for woodland habitats can be seen in figures 8.10 and 8.11. The mean critical load for managed coniferous woodland has increased from 1.851 keq ha^{-1} $year^{-1}$ to 2.057 keq ha^{-1} $year^{-1}$. For the managed broadleaved woodland the mean critical load has increased from 2.313 keq ha^{-1} $year^{-1}$ to 2.826 keq ha^{-1} $year^{-1}$.

As described in Chapter 5, the acidity Critical Load Function (CLF) is used to take account of the combined effects of sulphur and nitrogen deposition on acidification. The CLF is defined by three values, CLmaxS, CLmaxN, CLminN, which are determined by a range of parameters, including acidity critical loads (empirical values for soils, SMB values for woodland soils), non-marine base cation and chloride deposition, and uptake values for base cations, calcium and nitrogen. Broad changes to the mean CLmaxS, CLmaxN and CLminN for each habitat type are summarised in Table 8.3. The changes to the national maps are shown in figures 8.12 to 8.16 for CLmaxS, figures 8.17 to 8.21 for CLminN and figures 8.22 to 8.26 for CLmaxN.

Whether these changes result in the habitat being considered more or less sensitive to acid deposition depends on the incoming deposition (the amounts of S and N) in relation to the updated CLF. Each 1km grid square has a different CLF and each 5km grid square experiences a different deposition. Two examples of the possible effects of changing the CLF are given in figures 8.27 and 8.28. These examples are based on the mean previous and updated values of CLmaxS, CLminN and CLmaxN (Feb 2001 and Feb 2003) given in Table 8.3, using a fixed deposition (corresponding to the mean UK deposition values for 1995-97). Figure 8.27 for acid grassland shows that a small increase in mean CLminN and a decrease in mean CLmaxS and CLmaxN lead to a smaller CLF and a larger critical loads exceedance in 2003. By comparison, the decrease in mean CLminN and increase in mean CLmaxS and CLmaxN lead to a larger CLF for managed broadleaved woodland (Figure 8.28). However, the values for both 2001 and 2003 are above the mean deposition values, so no exceedance would occur with either set of values. For exceedance maps based on the CLF see Section 8.8 and the Addendum to this report.

8.7 Changes in nutrient nitrogen critical loads

Chapter 7 describes in detail the assignment of empirical nutrient nitrogen critical loads to the EUNIS habitat classes, and also the calculation of nutrient nitrogen critical loads for managed woodland habitats using the nitrogen mass balance equation. Table 8.4 summarises the new and old critical load values by habitat. The implications of changes in the values are not always clear on the national maps (Figures 8.29 to 8.33) because of the mapping class divisions used for consistency with maps of acidity critical loads.

Table 8.2 Summary of changes made to input parameters for acidity and nutrient nitrogen critical loads

Data set	**Broad Habitat**[1]	**Change**
Empirical acidity critical loads for soils	-	Revised 1km soils databases for GB
Acidity critical loads for peat soils	-	New method for calculation
Non-marine base cation deposition	All terrestrial habitats mapped	New data for 2000 used
Non-marine chloride deposition	All terrestrial habitats mapped	New data for 2000 used
Total calcium deposition	Coniferous woodland (managed) Broadleaved woodland (managed) unmanaged woodland	New data for 2000 used New data for 2000 used New data for 2000 used (not previously mapped)
Base cation uptake	Coniferous woodland (managed) Broadleaved woodland (managed)	New value 0.27 keq ha^{-1} $year^{-1}$; old value 0.25 keq keq ha^{-1} $year^{-1}$ New value for Ca-poor soils 0.315 keq ha^{-1} $year^{-1}$; old value 0.4 keq ha^{-1} $year^{-1}$ New value for Ca-rich soils 0.41 keq ha^{-1} $year^{-1}$; old value 0.85 keq ha^{-1} $year^{-1}$
Calcium uptake	Coniferous woodland (managed) Broadleaved woodland (managed)	New value 0.16 keq ha^{-1} $year^{-1}$; old value 0.12 keq keq ha^{-1} $year^{-1}$ New value for Ca-poor soils 0.195 keq ha^{-1} $year^{-1}$; old value 0.33 keq ha^{-1} $year^{-1}$ New value for Ca-rich soils 0.29 keq ha^{-1} $year^{-1}$; old value 0.7 keq ha^{-1} $year^{-1}$
Base cation weathering (ANC_w)	Coniferous woodland (managed) Broadleaved woodland (managed) unmanaged woodland	Revised 1km soils databases for GB Revised 1km soils databases for GB Revised 1km soils databases for GB (habitat not previously mapped)
Nitrogen uptake	Acid grassland Dwarf shrub heath Coniferous woodland (managed) Broadleaved woodland (managed) Bogs Montane	New value 1.14 kg N ha^{-1} $year^{-1}$; old value 1.0 kg N ha^{-1} $year^{-1}$ New value 0.5 kg N ha^{-1} $year^{-1}$, old value 4.0 kg N ha^{-1} $year^{-1}$ New value 2.94 kg N ha^{-1} $year^{-1}$; old value 7.0 kg N ha^{-1} $year^{-1}$ New value 5.88 kg N ha^{-1} $year^{-1}$; old value 7.0 kg N ha^{-1} $year^{-1}$ Value 0.5 kg N ha^{-1} $year^{-1}$ (habitat not previously mapped) Value 0.5 kg N ha^{-1} $year^{-1}$ (habitat not previously mapped)
Nitrogen immobilisation	All terrestrial habitats mapped[2]	Revised 1km soils databases for GB
Denitrification	All terrestrial habitats mapped[2]	Revised 1km soils databases for GB
Acceptable nitrate leaching	Coniferous woodland (managed) Broadleaved woodland (managed)	New value 4 kg N ha^{-1} $year^{-1}$; old value 6 kg N ha^{-1} $year^{-1}$ New value 3 kg N ha^{-1} $year^{-1}$; old value 6 kg N ha^{-1} $year^{-1}$

[1]The “broadleaved, mixed and yew woodland” broad habitat is separated into “broadleaved woodland (managed)” and “unmanaged (ancient and semi-natural) coniferous and broadleaved woodland” abbreviated to “unmanaged woodland” above; the latter includes Atlantic oak woods and unmanaged coniferous woodland (see Section 3.7.1).
[2]Nitrogen immobilisation and denitrification values for “standing open waters” and “rivers and streams” are based on catchment soils data, not 1km GB databases.

Table 8.3 Summary of changes in CLmaxS, CLminN, CLmaxN.

Critical load	Broad habitat[1]	Previous (Feb 2001) mean value (keq ha^{-1} $year^{-1}$)	Updated (Feb 2003) mean value (keq ha^{-1} $year^{-1}$)	Difference between previous and updated means[2]
CLmaxS	Acid grassland	1.046	0.824	Decrease 21.2%
	Calcareous grassland	2.768	3.920	Increase 29.4%
	Dwarf shrub heath	0.843	0.843	No change
	Coniferous woodland (managed)	1.828	1.965	Increase 7.0%
	Broadleaved woodland (managed)	2.101	2.660	Increase 21.0%
	Unmanaged woodland	Not mapped	3.243	
	Bogs	Not mapped	0.901	
	Montane	Not mapped	0.557	
	Standing open waters, rivers & streams	5.019	3.636	Decrease 27.6%
CLminN	Acid grassland	0.351	0.367	Increase 4.4%
	Calcareous grassland	1.214	0.889	Decrease 26.8%
	Dwarf shrub heath	0.580	0.851	Increase 31.8%
	Coniferous woodland (managed)	0.782	0.478	Decrease 38.9%
	Broadleaved woodland (managed)	0.747	0.663	Decrease 11.2%
	Unmanaged woodland	Not mapped	0.245	
	Bogs	Not mapped	0.343	
	Montane	Not mapped	0.318	
	Standing open waters, rivers & streams	0.288	0.307	Increase 6.2%
CLmaxN	Acid grassland	1.397	1.192	Decrease 14.7%
	Calcareous grassland	3.687	4.809	Increase 23.3%
	Dwarf shrub heath	1.424	1.695	Increase 16.0%
	Coniferous woodland (managed)	2.611	2.443	Decrease 6.4%
	Broadleaved woodland (managed)	2.599	3.323	Increase 21.8%
	Unmanaged woodland	Not mapped	3.488	
	Bogs	Not mapped	1.244	
	Montane	Not mapped	0.874	
	Standing open waters, rivers & streams	8.031	5.308	Decrease 33.9%

[1]The “broadleaved, mixed and yew woodland” broad habitat is separated into “broadleaved woodland (managed)” and “unmanaged (ancient & semi-natural) coniferous and broadleaved woodland” abbreviated to “Unmanaged woodland” above; the latter includes Atlantic oak woods and unmanaged coniferous woodland (see Section 3.7.1).

[2]An increase or decrease in the mean critical load values does not necessarily mean that all values for that habitat have increased or decreased, some may have increased in value and others decreased in value.

Table 8.4 Summary of changes in CLnutN

UPDATE (Feb 2003)			PREVIOUS (Feb 2001)		
Broad Habitat[1]	EUNIS classes	CLnutN values (kg N ha^{-1} $year^{-1}$)	Ecosystem	CLnutN categories	CLnutN values (kg N ha^{-1} $year^{-1}$)
Acid grassland	Dry acid & neutral closed grassland Moist or wet oligotrophic grassland	15 15	Acid grassland[#]	Neutral-acid species rich grassland Montane sub-alpine grassland Peat (bog)	25 12.5 10
Calcareous grassland	Semi-dry calcareous grassland	20	Calcareous grassland	Calcareous species-rich grassland	50
Dwarf shrub heath	Northern wet heaths Dry heaths	15 12	Heathland[##]	Lowland wet & dry heaths Species-rich heath/acid grassland Upland Calluna moor Arctic & alpine heath Peat (bog)	17 17 15 10 10
Bogs	Raised and blanket bogs	10		Mapped as part of acid grassland and heathland as above	
Coniferous woodland (managed)*	Coniferous woodland	8.9 – 13.9 mean 10.7	Coniferous woodland	Minimum of empirical value (13 kg N) or N mass balance (higher values)	13
Broadleaved woodland (managed)*	Broadleaved woodland	10.9 – 15.9 mean 12.3	Deciduous woodland	Minimum of empirical value (17 kg N) or N mass balance	15 – 17 mean 16.1
Unmanaged woodland	Broadleaved woodland (effects on ground flora)	12	Not mapped		
Broadleaved woodland (Atlantic oak woods)	Broadleaved woodland (effects on epiphytic lichens)	10	Not mapped		
Montane	Moss & lichen dominated summits	7	Not mapped	Not specifically mapped - areas included in acid grassland & heathland	
Supralittoral sediment	Shifting coastal dunes Stable dune grassland	15 15	Not mapped		

[1]The "broadleaved, mixed and yew woodland" broad habitat is separated into "broadleaved woodland (managed)", "broadleaved woodland (Atlantic oak woods)" and "unmanaged (ancient & semi-natural) coniferous and broadleaved woodland" (excluding Atlantic oak woods) abbreviated to "Unmanaged woodland" above; the latter includes unmanaged coniferous woodland (see Section 3.7.1).

* Nitrogen mass balance used (ie, Nu + Ni + Nde + Nl)

[#] Mean value on acid grassland map = 22.7 kg N ha^{-1} $year^{-1}$

[##] Mean value on heathland map = 14.9 kg N ha^{-1} $year^{-1}$

8.8 Resultant changes in critical load exceedance

The NFC has now used the updated critical load maps to calculate and map critical load exceedance, by comparison with national deposition maps for acidity and nitrogen. The resultant maps and statistics on the areas exceeding critical loads are published in the Addendum to this report on the UK NFC website (http://critloads.ceh.ac.uk).

The maps and statistics of exceedance are not submitted to the CCE, as they do not form part of the CCE's official call for data.

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