

# STATUS OF UK CRITICAL LOADS AND EXCEEDANCES JANUARY 1998

## PART 1 - CRITICAL LOADS AND CRITICAL LOADS MAPS

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# **STATUS OF UK CRITICAL LOADS AND EXCEEDANCES JANUARY 1998**

## **PART 1 - CRITICAL LOADS AND CRITICAL LOADS MAPS**

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### **EXECUTIVE SUMMARY**

Critical loads for acidity and nutrient nitrogen effects have been the basis for national and international deliberations on the controls for atmospheric emissions of sulphur and nitrogen pollutants. This report describes the methods and data used for producing national maps of critical loads for the UK. The latest data sets have been submitted to an international co-ordination centre for inclusion in European maps of critical loads; these will be used in future international agreements on emission controls in Europe.

The report describes the methods used for estimating acidity critical loads for a number of soil vegetation systems (acid grassland, calcareous grassland, heathland, coniferous and deciduous woodland). Both empirical and a simple mass balance approach have been used, each being applied appropriately to the different ecosystems. For freshwater systems, three different models for calculating critical loads are described. Methods for nutrient nitrogen critical loads for terrestrial systems are also summarised. Maps of critical loads for the different ecosystems and models, together with many of the data sets on which they are based are presented.

The critical loads function (CLF), a method of considering acidity critical loads in terms of both sulphur and nitrogen deposition, is explained and the calculation of the different components of the function described. In addition, the commonly used area percentile maps of critical loads, which indicate critical loads protecting proportions of ecosystems, are summarised with examples of UK maps.

### **1 INTRODUCTION**

The Critical Loads Mapping and Data Centre at CEH Monks Wood is designated as the UK National Focal Centre (NFC) for mapping critical loads. It is responsible for co-ordinating the critical loads mapping activities in the UK and compiling national critical loads data sets and maps from data supplied by UK experts. This report describes the latest critical loads data and information on how they have been prepared.

These data are particularly important as they have recently been submitted to the Co-ordination Centre for Effects (CCE) in the Netherlands who are producing European critical loads maps from national contributions. The European maps are being used, under the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP), to help develop a multi-pollutant, multi-effect Protocol for the control of sulphur and nitrogen pollutant emissions. The Protocol will address the problems of acidification, excess nutrient nitrogen and tropospheric ozone across Europe. The Protocol is in its final stages of negotiation and the recently submitted data are likely to remain fixed throughout the deliberations on future strategies and the setting of national obligations.

The European Union is also making use of the same critical loads database for its preparation of an Acidification Strategy and a Directive that will define national emission ceilings for the member states. The critical loads data that have been submitted are, therefore, of major political and scientific significance.

This report describes the methods used by experts in the UK to calculate critical loads of acidity and nutrient nitrogen. A critical load has been defined as " a quantitative estimate of an 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 ". For acidity, exceeding the critical load (termed exceedance) results in excess acidity in soils and waters which may be associated with biological effects. These may be due directly to acidity or, for example, an increase in dissolved aluminium concentrations caused by acid conditions. Effects of excess nitrogen as a nutrient can lead to leakage of nitrogen from soil systems or unwanted changes in species composition, for example, accelerated growth of some species at the expense of others.

The methods for calculating critical loads are based on agreed approaches that have been published in a Mapping Manual (UBA, 1996) by the CLRTAP's Task Force on Mapping. The methods have been adapted to make use of the national data sets that are available for producing our maps. Critical loads for acidity and also critical loads for excess nutrient nitrogen effects, have been calculated as required for the European mapping exercise.

For the UK, acidity critical loads have been calculated for six ecosystems: acid grassland, calcareous grassland, heathland, coniferous woodland, deciduous woodland and freshwaters. Nutrient nitrogen critical loads have been derived for all of these ecosystems except freshwaters.

Because acidity results from both sulphur and nitrogen pollutant deposition, the report describes the calculation of acidity critical loads in terms of either sulphur and/or nitrogen. These values together can be used to describe a Critical Loads Function (CLF) which has been used as a convenient way for estimating the excess deposition over the critical load, the exceedance. However, the calculation of exceedances will be addressed in a separate report.

## **2 CRITICAL LOADS OF ACIDITY -TERRESTRIAL ECOSYSTEMS**

Two methods are used in the UK for calculating acidity critical loads for terrestrial ecosystems: the empirical approach is used for estimates for non-woodland ecosystems and a simple mass balance (SMB) equation for woodland ecosystems. These methods are described in the sections below.

The SMB gives an "ecosystem" critical load which accounts for the acid tolerance of the plant through a chosen critical chemical criterion. This represents a philosophical shift from the empirical approach to the SMB approach. In combining empirical and SMB maps we are setting the critical ANC leaching term to zero for non-woodland systems and to a value greater than zero for woodland systems.

### **2.1 Empirical - soil-vegetation systems: acid grassland, calcareous grassland, heathland**

Empirical critical loads of acidity for soils have been assigned to each 1km grid square of the UK based upon the mineralogy of the dominant soil series present in the square (Hornung *et al.*, 1995a). The data are mapped in five classes representing ranges of critical loads values (Figure 1). Empirical critical loads for soils were the basis of the early critical loads maps for the UK.

We still use empirical acidity critical loads for non-woodland ecosystems (i.e. acid grassland, calcareous grassland, heathland). The alternative for soils, the current form of the Simple Mass Balance (SMB) equation, is only suitable for woodland ecosystems, given our current uncertainties in setting appropriate critical chemical limits for non-woodland systems.

For mapping the areas of acid grassland and heathland in Great Britain are defined from the CEH (ITE) Land Cover Map (Fuller *et al.*, 1994), where the areas of different land cover classes, combined to represent the ecosystem type, occupy more than 5% of a 1km grid square. 5% is assumed to represent a "significant" amount of ecosystems, and is intended to avoid problems from misclassifications, which can arise in satellite imagery for very small areas.

For calcareous grassland, the classes on the CEH (ITE) Land Cover Map for Great Britain do not distinguish this type from other managed grass, so the land cover classes were used in conjunction with species distribution data from the Biological Records Centre at CEH to define the ecosystem areas. Any remaining grid squares with critical loads  $\leq 0.5 \text{ keq H}^+ \text{ ha}^{-1} \text{ year}^{-1}$  were also removed from the data set, since it is unlikely that these are true areas of calcareous grassland. Again critical loads values were applied where the ecosystem occupied  $> 5\%$  of a 1km grid square.

The CEH (ITE) Land Cover Map does not include Northern Ireland and the Isle of Man. To define ecosystem areas for these regions alternative data sets were used. For Northern Ireland, the CORINE Level 3 map (CEC, 1992) was converted to the same classes as the CEH (ITE) Land Cover Map for Great

Britain using the software LUCID (Land Use Classifications Information and Documentation) version 1.0 (Wyatt *et al.*, 1990). For the Isle of Man, the CEH Countryside Information System (Howard *et al.*, 1994) was used in conjunction with LUCID to estimate the percentage areas of the five different terrestrial ecosystems (acid grassland, calcareous grassland, heathland, coniferous and deciduous woodland).

For the current UK maps the mid-range empirical critical load values from the ranges identified in Figure 1 are used (i.e. 0.1, 0.35, 0.75, 1.5 keq H + ha<sup>-1</sup> year<sup>-1</sup>), with the exception of those with the highest critical load where 4.0 keq H + ha<sup>-1</sup> year<sup>-1</sup> is used. This is consistent with calculations of the weathering rates for different soil types using the PROFILE model (Sverdrup *et al.*, 1990); these have shown that the calculated weathering rate falls mostly within the class ranges (Langan *et al.*, 1995) so a mid-range value gives a good estimate.

Maps showing the critical loads values for the areas of each ecosystem are given in Figures 2a, 2b and 2c.

## **2.2 Simple Mass Balance (SMB) model - coniferous and deciduous woodland ecosystems**

UK critical load experts agree that the current formulation of the SMB equation is appropriate for calculating critical loads of acidity for woodland ecosystems. Other European countries also use the method for calculating forest ecosystem critical loads. We believe that the application of the SMB equation (Appendix I) to non-forest systems needs further development and testing.

The SMB approach is based on balancing inputs of acidity to the ecosystem with sinks within the system and outputs. The critical load is calculated such that the balance between inputs, sinks and outputs ensures that a chosen critical chemical limit is not exceeded. The critical chemical limit is set to protect a biological indicator that is used as an indicator of the "health" of the forest (Hornung *et al.*, 1997). The critical chemical limit being applied in the UK is the ratio of calcium to aluminium in soil solution at which fine root damage occurs; a value of one is currently being used.

The methodology described in the Mapping Manual (UBA, 1996), and agreed by the Task Force on Mapping (TFM), recommends the use of "total" (non-marine plus marine) base cation (calcium plus magnesium) deposition (Figure 3a) in the SMB equation. However, using "total" base cation deposition and a base cation to aluminium ratio (Bc:Al) as the chemical criterion in the equation gives high critical loads across much of Britain, even in areas known to be sensitive to acidification. The Bc:Al ratio assumes that magnesium affords the same protection to plant roots as calcium but there are some doubts about this. The experimental work on plant growth which provides the basis of the SMB equation was focused almost entirely upon the ratio of calcium to aluminium (Ca:Al) concentrations in soil solution. The assumptions about the protection afforded by magnesium need checking experimentally and until then UK experts agree it is best to use the Ca:Al ratio, especially for high sea salt deposition areas such as the UK where magnesium deposition may be significant. Cronan & Grigal (1995) reviewed the use of Ca:Al as a chemical criterion for damage and concluded that a Ca:Al ratio of 1.0 was probably justified. The use of the SMB equation (Appendix 1) with a Ca:Al ratio requires that other parameters in the equation are also for calcium only instead of base cations, for example, "total" calcium deposition (Figure 3b) and estimates of calcium weathering.

For the UK, we have applied the SMB model separately for coniferous and deciduous woodland, in each case assuming that the woodland type covers the whole country (Figure 4a and Figure 5a). The critical load values for squares dominated by peat soils are replaced with the empirical critical load values calculated by the University of Aberdeen (Hornung *et al.*, 1995a; Smith *et al.*, 1993), as these are considered the most appropriate (Figure 4b and Figure 5b). The resulting maps are "masked" to present only those grid squares where the woodland type occupies more than 5% of the total grid area (Figure 4c and Figure 5c).

The sections below describe the derivation of base cation and calcium uptake values for woodland and the estimates of calcium weathering that are needed for the SMB calculations. The derivation of nitrogen uptake is also included in the following section, since the same approach is applied as for base cations and calcium uptake.

### **2.2.1 Estimates of base cation, calcium and nitrogen uptake for woodland ecosystems**

For the SMB equation using the Bc:Al ratio, net base cation uptake values must be determined. Therefore, if a Ca:Al ratio is used, values for calcium uptake must be estimated. However, base cation uptake estimates are still required for calculations related to critical loads exceedance, eg the maximum critical

load for sulphur (Section 5.1.1), as base cations neutralise acidity. Nitrogen uptake is also discussed in this section and the use of these values is described in Section 4.

Base cation (Bc = calcium + magnesium + potassium), calcium (Ca) and nitrogen (N) losses by uptake and removal in harvesting of forests and woodlands is calculated from the average volume increment (i.e. a measurement of yield) achieved by each species in Britain (FICGB, 1995). Average volume increments (in  $\text{m}^3 \text{ha}^{-1}$ ) are converted to amount removed in harvesting using species specific wood densities (Hamilton, 1975) and the Bc, Ca and N concentrations of timber from published studies (Emmett *et al.*, 1996; Stephens *pers com.*; Sverdrup *et al.*, 1995; Allen *et al.*, 1974). All calculations use the same equation:

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

Table 1 shows the results using data for deciduous and coniferous woodland at harvest; species differences are ignored. They are calculated using area-weighted average volume increments and wood densities, but BC, Ca and N concentrations for oak are used for deciduous woodland and Sitka spruce for coniferous woodland. Area-weighted average volume increments and wood densities are calculated from the average values for each species, with each species contributing to the average proportionally, depending on the proportion of UK coniferous or deciduous woodland which they represent (FICGB, 1995; Forestry Commission, 1996). We have assumed that only the stem is removed.

It is considered that species-specific data would significantly improve the accuracy of these estimates and good estimates of the Ca, magnesium (Mg), potassium (K) and N content in wood are currently being compiled. A difficulty with much of the published data is that nutrient contents are shown separately for wood and bark and in some cases for sapwood and heartwood without any indication of the relative proportions of each. Bark is normally removed from the forest along with wood at harvest and thus nutrient content data which includes both components have been used (Table 1).

For Sitka spruce, data from two studies which show similar wood nutrient contents (Emmett *et al.*, 1996; Stephens *pers comm.*) were used. However, additional studies which show bark and wood values separately (Minocha & Shortle, 1993; Carey, 1980; Miller *et al.*, 1993) gave figures which compared well with those used in the calculations. For oak, nitrogen data were taken from Allen *et al.* (1974) and were the only published UK data that could be found.

In contrast to the conifers, and to Sitka spruce in particular, the Ca content of oak wood appears to be very variable, reflecting the range of soils on which the species grows and thus a wide tolerance to different Ca availability in soil. Brown (1974), for example, showed that annual Ca removal in increment varied by about a factor of ten between a base poor site (Bogle Crag, Cumbria) and a base rich site (Meathop Wood, Cumbria), and considered that "luxury consumption" of Ca occurs where soil availability is high. In addition, calcium is the dominant of the three base cations so that variability in its concentration in wood will also give a large range in base cation uptake. Thus for Ca and BC uptake by deciduous trees, upper limit and lower values were used from Sverdrup *et al.* (1995) and Allen *et al.* (1974) respectively.

To apply the above uptake values for deciduous woodland, areas of calcium rich and calcium poor soils were defined from a 1km map which divides soils into three sensitivity classes depending on their base saturation and pH (Hornung *et al.*, 1995b). Those with a pH below 4.5 and base saturation <20% were classified as Ca-poor and those with higher pH and base saturation values as Ca-rich soils (Figure 6).

The above classification has been used for the UK, however, this has led to some counter-intuitive results, with higher critical loads in some areas of Ca-poor soils and lower critical loads in some areas of Ca-rich soils. This is a consequence of taking a single uptake value for Ca-poor or Ca-rich soils and applying them at the national scale. Further experimental work in the next three years will provide additional data to improve the uptake estimates for Ca-poor and Ca-rich soils. Critical loads estimates will also be improved as a result.

In Britain, average volume increments vary considerably between conifer species and less between the main deciduous species. Similarly wood nutrient content data vary considerably between species. Consequently moving to species-specific values of nutrient removal would significantly improve the

estimates shown in Table 1. Although there is significant regional variability in average yields (Nicholls, 1981; Kupiec & Philips, 1988) this tends to be accommodated by species choice so that individual species are generally planted where they grow well. Thus moving to average yields based on local, Forestry Commission Conservancy or regional yield tables is unlikely to improve the accuracy of data over species-specific values based on national average yields.

For a number of reasons the forecasts of volume production in Britain are not achieved (Kupiec & Philips, 1988). In the uplands, windblow, premature felling and failure to remove thinnings are all contributing factors. While in the lowlands the management of woods for objectives other than timber production (conservation, game, amenity etc) is an additional reason for potential volume production not being achieved. Even where management is according to normal practice for optimisation of volume increment, increment is commonly not harvested. On a European basis only 70% of annual volume increment is removed in harvesting (UNECE/FAO, 1996). Data not corrected for the failure to realise timber potential (ie failure to manage for and harvest average volume increment) represent a considerable overestimate of timber removal and therefore the worst case for base cation removal and the best case for nitrogen removal. Data with "best guess" corrections for timber potential realised have therefore been added to Table 1. These assume that 50% of the average volume increment is achieved and removed at harvest for deciduous trees and 75% for conifers. Published data for annual UK wood production (FICGB, 1995) indicate that even with 50% and 75% corrections we have overestimated base cation, calcium and nitrogen removal through an optimistic estimate of timber production for most years. The data without the correction factors were used in the final SMB critical load because these represent the theoretical maximum removal figures.

### 2.2.2 Estimates of calcium weathering rates

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

$$CL(A) = ANC_w - ANC_{le(crit)}$$

where  $ANC_w$  = weathering of base cations  
and  $ANC_{le(crit)}$  = critical leaching of base cations.

In the UK, the mid-range values of the empirical critical loads (as defined in Section 2.1) are used for  $ANC_w$  (Figure 7a). In this UK figure, peat squares are shown with zero values but are replaced with empirical critical load values in the final maps, see Section 2.2.

The calculation of  $ANC_{le(crit)}$  also uses the weathering rate ( $ANC_w$ ); for the SMB based on the BC:Al ratio the same "mid-range empirical"  $ANC_w$  values were used ie base cation weathering. However, for the SMB using the Ca:Al ratio, the weathering component of  $ANC_{le(crit)}$  must be re-calculated for Ca weathering only. To achieve this, "calcium correction" values have been assigned to each soil series (by M. Hornung, P. Loveland & S. Langan) previously used to determine empirical critical loads. The "calcium correction" values were derived from published geochemical data used to calculate the ratio of Ca: (Ca + Mg + K). These are assigned to each 1km square of Britain for the dominant soil series. Estimates for England and Wales and for Northern Ireland were prepared by the Soil Survey and Land Research Centre (SSLRC) and for Scotland by the Macaulay Land Use Research Institute (MLURI). Calcium weathering ( $Ca_w$ ) values (Figure 7b) were then calculated as:

$$Ca_w = ANC_w * \text{calcium correction value}$$

These values are used to calculate  $ANC_{le(crit)}$  using the SMB Ca:Al equation (Appendix I).

## 3 CRITICAL LOADS OF ACIDITY - FRESHWATER ECOSYSTEMS

Two empirical models have been used in the UK to calculate both sulphur and acidity critical loads for freshwaters, the Diatom model and the Steady-State Water Chemistry (SSWC) model. In addition, to provide the data needed for the CLRTAP multi-pollutant, multi-effect Protocol, the First-order Acidity Balance (FAB) model has been used to calculate the critical loads of sulphur and nitrogen of the Critical Loads Function (Section 5).

### 3.1 Empirical models

The Diatom and Steady-State Water Chemistry models have been described in detail elsewhere (CLAG, 1995; Allott *et al.*, 1995; Harriman *et al.*, 1995; Hornung *et al.*, 1997). Maps have been produced from

calculations using data from a survey of 1573 waters from single lakes or headwater streams, judged to be the most sensitive (in terms of acidification) in each 10km grid square of the UK. In less sensitive areas (e.g. south-east England) the sampling was generally from one site in a 20km grid square. It should be noted that the critical load value mapped for a given 10km grid square is only applicable to the waterbody or stream sampled and cannot be applied to other waters within the same grid square.

### 3.1.1 Diatom model

This model is based on a dose-response relationship between total acid deposition and changes in diatom composition, taking into account variations in site sensitivity as represented by water calcium concentrations. Acidification is indicated by a shift to a more acid-tolerant diatom flora. The model is based on the point of onset of acidification and is therefore said to set a "base critical load" for a site (CLAG, 1995). The critical loads map produced from the national survey is shown in Figure 8a; non-sensitive areas not sampled in the survey are assigned to the class of highest critical loads.

### 3.1.2 Steady-State Water Chemistry model

This model is based on the principle that excess base cation run-off reflects the net weathering processes in a catchment. The critical load calculated by this method should be equal to, or greater than, the acidic input. By setting the Acid Neutralising Capacity (ANC) at a biologically significant level, a critical load can be calculated for any target organism. In the UK an ANC value of zero is used, indicating a 50% probability of damage to brown trout populations (CLAG, 1995). The critical loads map derived from this model is shown in Figure 8b and, as for the Diatom model, non-sensitive areas not sampled in the survey are assigned to the class of highest critical loads.

## 3.2 First Order Acidity Balance (FAB) Model

The FAB model applied to UK freshwaters (Curtis *et al.*, 1998) is based on previous work by Downing *et al.* (1993), Henriksen *et al.* (1993) and Kämäri *et al.* (1992). It enables estimates of the combined critical loads for nitrogen and sulphur for aquatic ecosystems to be calculated based on a charge balance. It takes into account catchment specific data such as deposition, areas of forest, and lake to catchment ratios.

The acidity balance for the FAB model is as follows:

$$N_{dep} + S_{dep} = fN_u + (1-r)(N_i + N_{de}) + rN_{ret} + rS_{ret} + BC_l - ANC_l$$

Where:

$N_{dep}$  = nitrogen deposition

$S_{dep}$  = sulphur deposition

$f$  = fraction of forest in the catchment

$r$  = lake to catchment ratio

$N_u$  = net growth uptake of N by coniferous forests in the catchment

$N_i$  = nitrogen immobilisation in the catchment soils

$N_{de}$  = nitrogen lost by denitrification

$N_{ret}$  = nitrogen retention in the lake

$S_{ret}$  = sulphur retention in the lake

$BC_l$  = base cation leaching from the catchment

$ANC_l$  = ANC leaching from the catchment

Catchment boundaries for the 1445 sites sampled in Great Britain have been defined and digitised to calculate:

- (i) Catchment areas (ie freshwater ecosystem areas);
- (ii) lake catchment ratios;
- (iii) the area of forestry in the catchment (using the CEH (ITE) Land Cover Map (Fuller *et al.*, 1994));
- (iv) catchment-weighted estimates of sulphur and nitrogen deposition from 1992-94 mean 20km data;
- (v) catchment-weighted estimates of runoff from 1km data;
- (vi) catchment-weighted estimates of nitrogen immobilisation and denitrification, based on the default values given in Hall *et al.*, (1997) and Appendices IV and V.

Net growth uptake of nitrogen was calculated by forest fraction multiplied by a fixed value of 4 kg N ha<sup>-1</sup> year<sup>-1</sup> (Curtis *et al.*, 1998). In-lake sulphur and nitrogen retention values were calculated from mass transfer coefficients taken from the Mapping Manual (UBA, 1996). Pre-industrial base cation leaching values were obtained with the Steady State Water Chemistry (SSWC) model. The ANC limit was set to

zero (as in the SSWC model for UK waters) based on a 50% probability of damage to brown trout populations.

The FAB model has been applied to all 1445 sites throughout Great Britain; it has been used to generate the critical loads of sulphur and nitrogen for the Critical Loads Function (see Section 5). Soils data are not currently available to apply the model to the sites in Northern Ireland.

#### 4 CRITICAL LOADS OF NUTRIENT NITROGEN - TERRESTRIAL ECOSYSTEMS

Critical loads for nutrient nitrogen ( $CL_{nut}(N)$ ) are calculated using two different methods, empirical and mass balance (UBA, 1996). In the UK, we have used empirical, mass balance or both of these approaches to calculate nutrient nitrogen critical loads for each of the terrestrial ecosystems for which acidity critical loads have been derived.

##### 4.1 Empirical critical loads of nutrient nitrogen

Empirical critical loads have been defined for a number of different vegetation types at international UNECE workshops (Grennfelt & Thornelöf, 1992; Hornung *et al.*, 1995c; UBA, 1996). The latest values are given in Appendix II.

To define empirical critical loads for four of the terrestrial ecosystems (acid grassland, calcareous grassland, heathland, and deciduous woodland) mapped for acidity stepwise methods have been developed. These make use of the 1km CEH (ITE) Land Cover Map (Appendix III), the National Vegetation Classification (Rodwell, 1991a, 1992b, 1992, 1994) and Biological Records Centre data; the sequential processes ensure that the lowest critical loads value for any appropriate ecosystem present in a 1km square is used. The procedures are detailed below.

Defining critical loads of nutrient nitrogen for acid grassland (Figure 9a):

- (i) Select the areas where acid grassland land cover classes (ie CEH (ITE) Land Cover classes 8, 9, 10, 12, 17, 19, 23, 24) occupy > 5% of a 1km square, and apply a critical load of 25 kg N ha<sup>-1</sup> year<sup>-1</sup> (i.e. the value for neutral-acid species rich grassland).
- (ii) If montane sub-alpine grassland is present, set the critical load for the 1km square to 12.5 kg N ha<sup>-1</sup> year<sup>-1</sup>.
- (iii) If the 1km square is dominated by peat soils and the dominant land cover is acid grassland, set the critical load to 10 kg N ha<sup>-1</sup> year<sup>-1</sup>.

Defining critical loads of nutrient nitrogen for calcareous grassland (Figure 9b):

- (i) Select the areas where calcareous grassland land cover classes (i.e. CEH (ITE) Land Cover classes 6, 7) occupy > 5% of a 1km square. If species-rich calcareous grassland (more or less N & P limited) is present (according to BRC records) apply a critical load of 50 kg N ha<sup>-1</sup> year<sup>-1</sup> (i.e. the value for calcareous species-rich grassland).

Defining critical loads of nutrient nitrogen for heathland (Figure 9c):

- (i) Select the areas where heathland land cover classes (ie CEH (ITE) Land Cover classes 5, 11, 13, 25) occupy > 5% of a 1km square, and apply a critical load of 17 kg N ha<sup>-1</sup> year<sup>-1</sup> (i.e. the value for lowland wet + dry heaths and species rich heath/acid grassland)
- (ii) If upland *Calluna* moorland is present, set the critical load for the 1km square to 15 kg N ha<sup>-1</sup> year<sup>-1</sup>.
- (iii) If arctic and alpine heath is present, set the critical load for the 1km square to 10 kg N ha<sup>-1</sup>.
- (iv) If the 1km square is dominated by peat soils and dominant land cover is heathland, set the critical load for 1km square to 10 kg N ha<sup>-1</sup> year<sup>-1</sup>.

Defining critical loads of nutrient nitrogen for deciduous woodland:

- (i) Select the areas where deciduous woodland land cover classes (i.e. CEH (ITE) Land Cover classes 14 and 15) occupy > 5% of a 1km square, set the critical load for 1km square to 17 kg N ha<sup>-1</sup> year<sup>-1</sup> (i.e. the value for changes in ground flora).

##### 4.2 Mass balance critical loads of nutrient nitrogen for woodland ecosystems

The mass balance approach balances nitrogen inputs and outputs at equilibrium and can, in principle, be used to calculate critical loads of nutrient nitrogen for any terrestrial ecosystem to prevent N-accumulation, nutrient imbalances and N-leaching (UBA, 1996). In the UK, this method is only used for calculating nutrient nitrogen critical loads for woodland ecosystems. However, this section considers input variables to the equation for other ecosystems, since they are required for calculating other critical



loads e.g. the minimum critical load of nitrogen (Section 5.1.2). The mass balance equation for  $CL_{nut}(N)$  is:

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

where

$N_u$  = nitrogen uptake

$N_i$  = nitrogen immobilisation

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

$N_{de}$  = denitrification

The derivation of nitrogen uptake values for woodland ecosystems is described above in Section 2.2.1. For other ecosystems, UK experts have agreed upon 0.07 keq ha<sup>-1</sup> year<sup>-1</sup> (1 kg N ha<sup>-1</sup> year<sup>-1</sup>) for acid grassland, 0.714 keq ha<sup>-1</sup> year<sup>-1</sup> (10 kg N ha<sup>-1</sup> year<sup>-1</sup>) for calcareous grassland and 0.29 keq ha<sup>-1</sup> year<sup>-1</sup> (4 kg N ha<sup>-1</sup> year<sup>-1</sup>) for heathland.

Nitrogen immobilisation and denitrification values are dependent on soil type (Appendices IV & V). Values for  $N_i$  (Figure 10a) are 0.07 and 0.21 keq ha<sup>-1</sup> year<sup>-1</sup> (i.e. 1 and 3 kg N ha<sup>-1</sup> year<sup>-1</sup>) and for  $N_{de}$  (Figure 10b) 0.07, 0.14 and 0.28 keq ha<sup>-1</sup> year<sup>-1</sup> (i.e. 1, 2 and 4 kg N ha<sup>-1</sup> year<sup>-1</sup>).

Nitrogen leaching is ecosystem specific. The UK has used a value of 0.14 keq ha<sup>-1</sup> year<sup>-1</sup> (2 kg N ha<sup>-1</sup> year<sup>-1</sup>) for acid grassland, calcareous grassland and heathland and a value of 0.43 keq ha<sup>-1</sup> year<sup>-1</sup> (6 kg N ha<sup>-1</sup> year<sup>-1</sup>) for woodland ecosystems.

The rates for nitrogen immobilisation and denitrification are for long-term sustainability; they may be, and often are, lower than values observed currently. Further information on the recommended range of values for  $N_u$ ,  $N_i$ ,  $N_j$  and  $N_{de}$  for the ecosystems being considered in Europe is given in Chapter 5 of the Mapping Manual (UBA, 1996).

The mass balance model is applied separately for coniferous and deciduous woodland, assuming that the woodland type covers the whole country. Additionally, for deciduous woodland the lowest critical loads value, either empirical or mass balance derived, is applied to each grid square. As previously, the resulting maps are then "masked" to present only those grid squares where the woodland type occupies more than 5% of the total grid area (Figures 11a & 11b).

## 5 THE CRITICAL LOADS FUNCTION

The discussions under the CLRTAP on a multi-pollutant, multi-effect Protocol are considering abatement strategies for both sulphur and nitrogen emissions. For this, the concept of a so-called Critical Loads Function (CLF) has been developed which can be used to consider the acidifying effects of sulphur and nitrogen deposition simultaneously. This enables estimates of sulphur and nitrogen deposition from atmospheric transport model calculations (eg as used in integrated assessment models) to be compared directly with "an acidity critical load".

In its simplest form, an acidity critical load may be defined graphically on a sulphur-nitrogen deposition plot by a 45 degree diagonal line. This intercepts the x-axis (representing nitrogen deposition) and y-axis (representing sulphur deposition) at chemically equivalent points, each representing the critical load for acidity in terms of nitrogen or sulphur only (Figure 12a). Each point along the diagonal line represents the critical load in terms of some deposition of sulphur and nitrogen.

To allow for nitrogen removal processes from the soil, the simple diagonal line is shifted along the nitrogen axis to increase the nitrogen values across the entire acidity CLF (Figure 12b). More nitrogen can then be deposited before the acidity critical load is exceeded. There are no similar removal processes that need to be considered for sulphur.

It is possible to incorporate in the CLF a critical load for the nutrient effects of nitrogen, by introducing a nitrogen limit of this value across all sulphur deposition values (i.e. a straight line parallel to the sulphur deposition axis, Figure 12c). However, in line with agreements in the CLRTAP and consistent with the work of the Task Force on Integrated Assessment Modelling (TFIAM), we define an acidity CLF (Figure 12b) for the UK. The CLF is used to look at the acidifying effects of nitrogen and sulphur only. Nutrient nitrogen critical loads (and their exceedances) are examined separately (Section 4 above).

The acidity CLF can be used in conjunction with measured, or modelled estimates of sulphur and nitrogen deposition (current and future), to ascertain deposition reductions required to achieve ecosystem protection (i.e. non-exceedance of critical loads). One method of visualising this is to identify six "regions" in relation to the CLF and identify the pollutant(s) that need(s) to be reduced to meet critical loads (Figure 13). The CLF can also be used to calculate the total amount of acidifying deposition that needs to be reduced in order to obtain protection. This will be described in detail in Part 2 of this report (Exceedance of Critical Loads).

### 5.1 Defining the "nodes" on the CLF

The intercepts of the CLF on the sulphur and nitrogen axes define the "maximum" critical loads of sulphur and nitrogen (i.e. the critical load for either sulphur or nitrogen alone). The nitrogen removal processes define a "minimum" critical load for nitrogen (Figure 12b). It is these values for each ecosystem considered nationally, together with the data used to calculate them, that are required for the integrated assessment modelling activities of the TFIAM which form the basis of the CLRTAP negotiations. The UK is currently submitting data for acid grassland, calcareous grassland, heathland, coniferous woodland, deciduous woodland and freshwater ecosystems. The sections below describe the "nodes" of the CLF (i.e. the maximum and minimum critical loads) and how they are derived. The equations used to calculate these critical loads differ for the terrestrial and the freshwater ecosystems; for soils they are defined using acidity critical loads whereas for freshwaters they are defined directly from the FAB model.

#### 5.1.1 Maximum critical load of sulphur $CL_{max}(S)$

The maximum critical load of sulphur ( $CL_{max}(S)$ ) is the critical load for acidity assuming sulphur only contributes to acidification, ie when nitrogen deposition is zero. It is defined in order to take account of any buffering effects of base cation deposition and removal of base cations by vegetation growth.

*(i) Defining  $CL_{max}(S)$  for terrestrial soil-vegetation ecosystems (acid grassland, calcareous grassland, heathland, coniferous and deciduous woodland)*

For soil-vegetation systems, the maximum critical load of sulphur is based upon the acidity critical load values but takes into account net base cation deposition to the soil system and base cation removal from the system:

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

where

$CL(A)$  = critical load of acidity (empirical or SMB)

$BC_{dep}$  = non-marine base cation deposition less non-marine chloride deposition

$BC_u$  = base cation uptake by vegetation

The critical loads of acidity are those calculated using the empirical or simple mass balance methods described in Section 2. Non-marine chloride is assumed to be associated with hydrochloric acid deposition and is therefore acidifying. For practical purposes it is therefore subtracted from the base cation deposition, since only sulphur and nitrogen deposition are considered for emission controls.

The chloride deposition values used for the UK are estimates of measured chloride deposition for 2010. This is consistent with the conclusions from discussions in the Task Force on Mapping where the end date of the Second Sulphur Protocol was identified as a suitable year for estimating chloride deposition. Non-marine chloride deposition results from the burning of some fossil fuels and can be modelled as for sulphur and nitrogen. However, the chloride deposition values were estimated by multiplying the current (1992-94) measured values by the ratio of modelled chloride deposition for 2010 to current modelled chloride deposition; this provides a better estimate than using modelled data alone.

For base cation deposition, current (1992-94) measured non-marine data are used since no estimates of future base cation deposition are available i.e. base cation deposition is assumed to remain constant. The appropriate deposition estimates are applied to the calculations for each ecosystem, i.e. low vegetation estimates for non-woodland ecosystems and woodland estimates for the woodland ecosystems. The estimates for base cation uptake are also ecosystem specific: zero for acid grassland and heathland ecosystems and  $0.222 \text{ keq ha}^{-1} \text{ year}^{-1}$  for calcareous grassland. The derivation of the values applied to the

woodland ecosystems is described in Section 2.4. The  $CL_{max}(S)$  values for these five terrestrial ecosystems are shown in Figure 14 and Figure 15.

(i) *Defining  $CL_{max}(S)$  for freshwater ecosystems*

The calculation of  $CL_{max}(S)$  for freshwaters is related to the critical leaching of base cations and the sulphur retention in the lake:

$$CL_{max}(S) = L_{crit} / (1 - r_s)$$

where  $r_s$  = modelled value for sulphur retained in the lake (calculated from lake depth, lake retention time, runoff and the net mass transfer coefficient for sulphur)

and  $L_{crit}$  = critical leaching flux of acid anions, calculated as:

$$L_{crit} = Q * ([BC * J_0] - [ANC]_{limit})$$

where  $Q$  = catchment-weighted runoff derived from 1km data

$[BC * J_0]$  = pre-acidification base cation concentration derived from the SSWC model

$[ANC]_{limit}$  = ANC leaching from catchment

$[ANC]_{limit}$  is set to zero and  $r_s$  to 0.5 metres year<sup>-1</sup> (see section 3.2).

This calculation of  $CL_{max}(S)$  takes account of base cation deposition in a less direct way than for terrestrial ecosystems, as  $BC_{dep}$  is included in the calculation of  $[BC * J]$ . The  $CL_{max}(S)$  values for freshwaters are shown in Figure 16.

**5.1.2 Minimum critical loads of nitrogen ( $CL_{min}(N)$ )**

$CL_{min}(N)$  is the deposition independent critical load of acidity solely due to nitrogen removal processes in the soil (nitrogen uptake and immobilisation).

(i) *Defining  $CL_{min}(N)$  for terrestrial soil-vegetation ecosystems (acid grassland, calcareous grassland, heathland, coniferous and deciduous woodland)*

$$CL_{min}(N) = N_u + N_i$$

where  $N_u$  = nitrogen uptake

$N_i$  = nitrogen immobilisation

Methods for the derivation of values for  $N_u$  and  $N_i$  have already been described in Sections 2.2.1 and 4.2 and values are mapped in Figures 10a and 10b.

(ii) *Defining  $CL_{min}(N)$  for freshwater ecosystems*

For freshwaters, nitrogen uptake is multiplied by the fraction of forestry in the catchment. The equation also includes the lake:catchment ratio (R) and denitrification:

$$CL_{min}(N) = fN_u + (1-R)(N_i + N_{de})$$

$N_i$  and  $N_{de}$  are catchment-weighted values according to the area of different soil types present in the catchment.

**5.1.3 Maximum critical loads of nitrogen ( $CL_{max}(N)$ )**

The maximum critical load of nitrogen is the critical load of acidity assuming only nitrogen contributes to acidification, ie when sulphur deposition is zero. It is therefore equivalent to the maximum critical load for sulphur but also allows for nitrogen removal processes.

(i) *Defining  $CL_{max}(N)$  for terrestrial soil-vegetation ecosystems (acid grassland, calcareous grassland, heathland, coniferous and deciduous woodland)*

For these ecosystems  $CL_{max}(N)$  is defined by  $CL_{min}(N)$  and  $CL_{max}(S)$ :

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

The methods for deriving the inputs required to calculate  $CL_{max}(N)$  are described above. Maps showing the  $CL_{max}(N)$  values for the five terrestrial ecosystems are shown in Figure 17 and Figure 18.

(ii) *Defining  $CL_{max}(N)$  for freshwater ecosystems*

The calculation of  $CL_{max}(N)$  for freshwaters is related to the critical leaching of base cations and nitrogen retention in the lake:

$$CL_{max}(N) = CL_{min}(N) + (L_{crit} / 1 - r_N)$$

where  $L_{crit}$  = critical leaching flux of acid anions (see section 5.1.1 above)  
 $r_N$  = modelled value nitrogen retained in the lake ie, the fraction of net inputs to the lake after denitrification, immobilisation and uptake.

The calculated  $CL_{max}(N)$  values for freshwaters are shown in Figure 19.

**5.1.4 Including nutrient nitrogen into the CLF**

The effects of nutrient nitrogen can be incorporated into a CLF so that both the acidifying and nutrient effects of nitrogen and the acidifying effects of sulphur can be examined simultaneously (Figures 12c & 12d). While this methodology is no longer used for developing abatement strategies in Europe (see Section 5), for completeness the derivation of the nodes on the CLF representing the "minimum critical load of sulphur" and the "limiting critical load of nitrogen" are given below.

*Minimum critical load of sulphur ( $CL_{min}(S)$ ):*

$$CL_{min}(S) = CL_{max}(S) - N_{le(ace)}$$

Values for acceptable nitrogen leaching ( $N_{le(ace)}$ ) are given in Section 4.2.

*Limiting critical load of nitrogen ( $CL(N)_{lim}$ ):*

The maximum critical load of nitrogen is re-defined as the "limiting" critical load of nitrogen, when the nutrient effects of nitrogen are included in the CLF (Figure 12d):

$$CL(N)_{lim} = \text{minimum} (CL_{max}(N), CL_{nut}(N))$$

ie the smallest value of  $CL_{max}(N)$  or  $CL_{nut}(N)$  is used instead.

**6 PERCENTILE CRITICAL LOADS**

The above sections have all described the derivation of critical loads and the data and maps for individual ecosystems. These can be used separately, for example, with the CLF to look at exceedances for a single ecosystem. However, for the national and European scale critical loads and exceedance maps it is more useful to combine the data for all ecosystems. Certain methods have therefore been adopted by the CLRTAP for developing protocols and by the EU for its acidification strategy. These have been based upon a statistical interpretation of all critical loads within a specified grid area. For Great Britain and Northern Ireland the grid size may be the national 1km, 10km, or 20km squares and for Europe the EMEP 50km or 150km grid squares (Posch *et al.*, 1997b). This percentile area critical loads approach is described below.

In the statistical approach (Posch *et al.*, 1997b), the critical load values for all ecosystems and their associated areas are ranked from lowest to highest, for each grid square. This provides the basis for a frequency distribution of critical loads based upon the ecosystem areas. Area percentiles may then be identified from the frequency distribution of the critical load values that will be associated with levels of ecosystem protection within the grid square. For example, a 5-percentile critical load will be the value that will protect 95% of the total ecosystem area within each grid square. The percentiles commonly mapped are the 5- and 50-percentile. More values are used for the purposes of integrated assessment modelling; 29 different percentiles are commonly calculated: 1<sup>-10</sup> in steps of 1, and 10<sup>-100</sup> in steps of 5.

A map of the 5-percentile critical loads of acidity is used as the national map for the UK (Figure 20). It combines the empirical critical loads for acid grassland, calcareous grassland and heathland with simple mass balance derived critical loads for coniferous and deciduous woodland. At the present time, this map

does not include acidity critical loads for freshwaters, which are mapped at a different scale; catchment boundaries are being used to generate area-weighted freshwater critical loads at the 1km scale, and these will be incorporated into the national 1km map. The current map differs from the earlier national map (Figure 1) in that non-sensitive areas (primarily agricultural and urban areas) are no longer mapped. Other examples of 5-percentile critical loads at 1km resolution are shown in Figures 21a, 21b and 21c.

For integrated assessment modelling and the development of abatement strategies for emissions of sulphur and nitrogen, maps at the EMEP 150km scale are being used. These include national data from participating countries to the Convention on Long-Range Transboundary Air Pollution.

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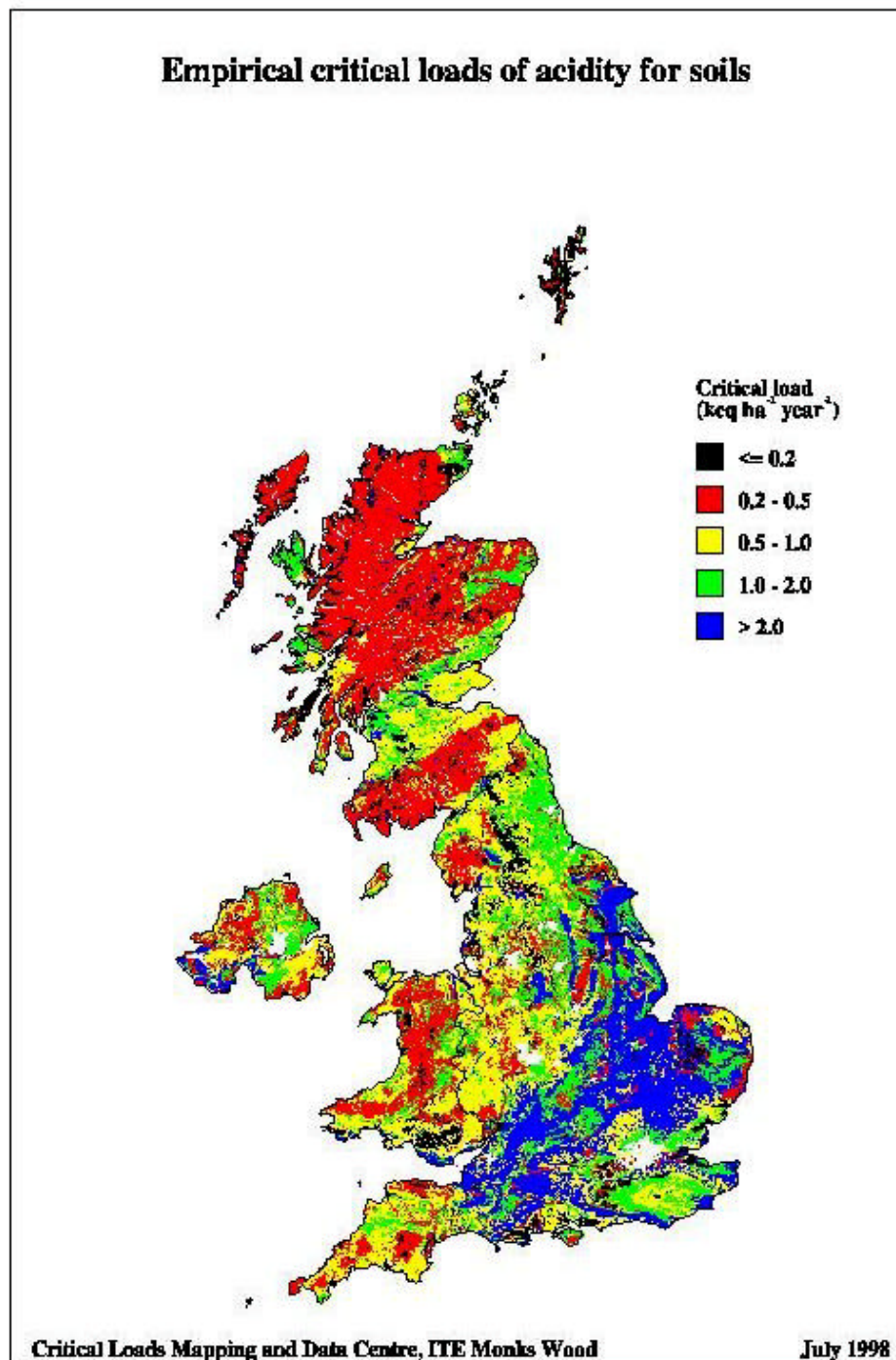
**TABLE 1**

Base cation, calcium and nitrogen uptake values for coniferous and deciduous woodland at harvest. Values are shown assuming that potential yields are achieved and also with a correction factor to account for harvests being below potential ("estimated" yield).

Woodland type and yield achieved	Uptake values in keq ha <sup>-1</sup> year <sup>-1</sup>		
	base cations	calcium	nitrogen
<b>Potential yield:</b>			
Conifers	0.253	0.117	0.279
Deciduous Ca-rich soils	0.613	0.516	0.278
Deciduous Ca-poor soils	0.171	0.076	0.278
<b>Estimated yield:</b>			
Conifers	0.187	0.087	0.146
Deciduous Ca-rich soils	0.306	0.258	0.136
Deciduous Ca-poor soils	0.085	0.034	0.136

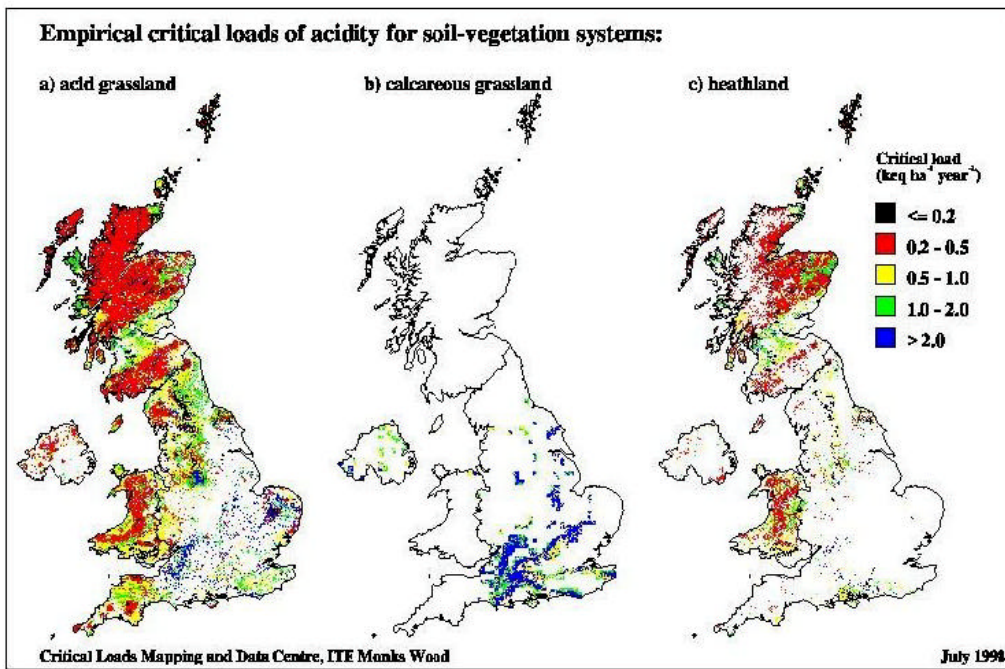


Figure 1



Data acknowledgment: UK Critical Loads Network (CLONE).

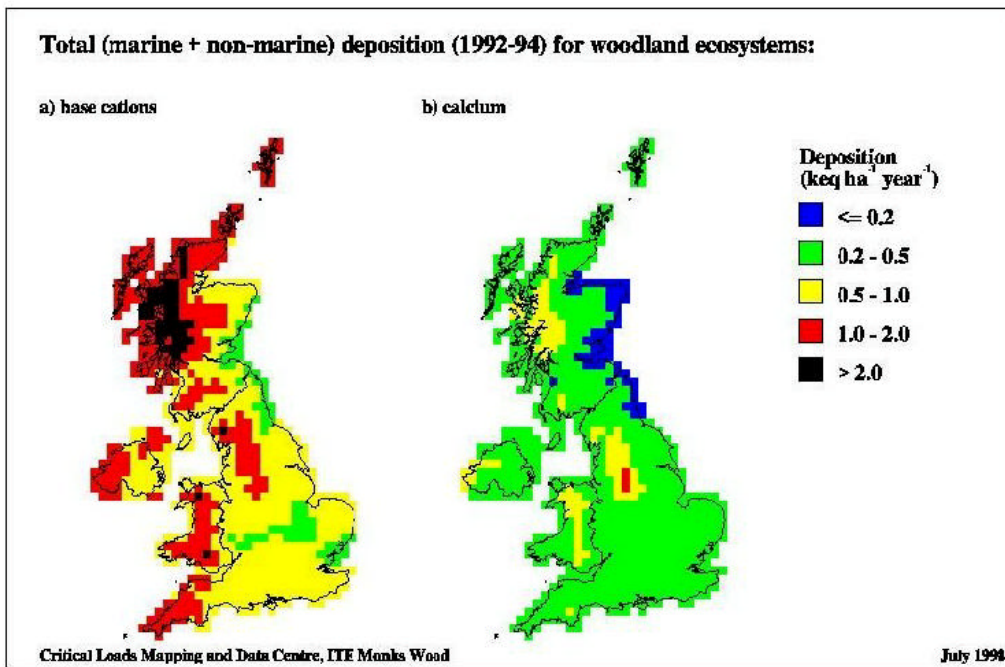
NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.



Data acknowledgement: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

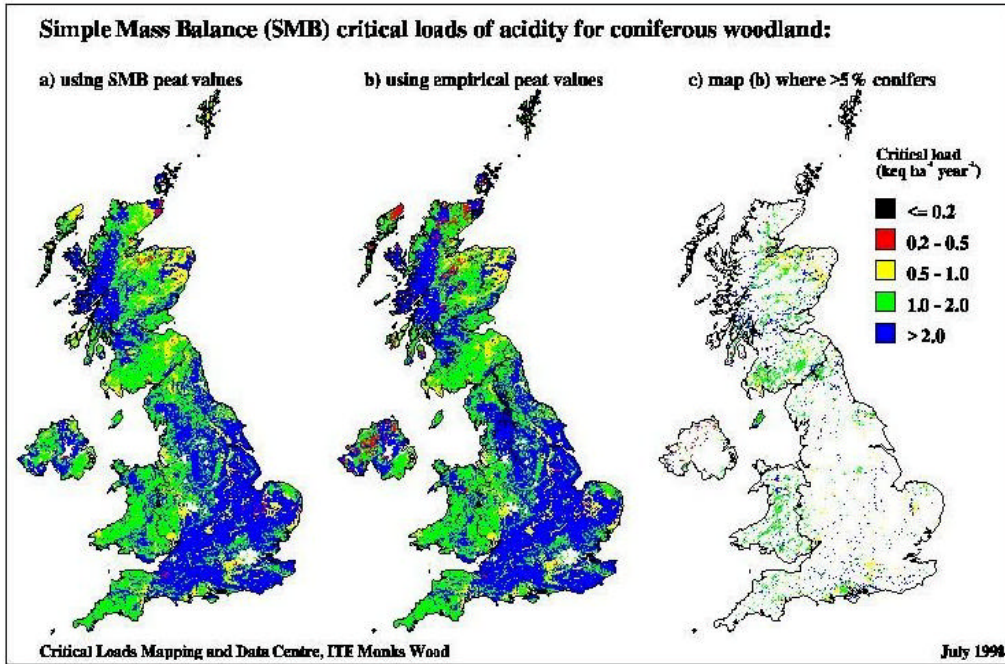
Figure 2



Data acknowledgement: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

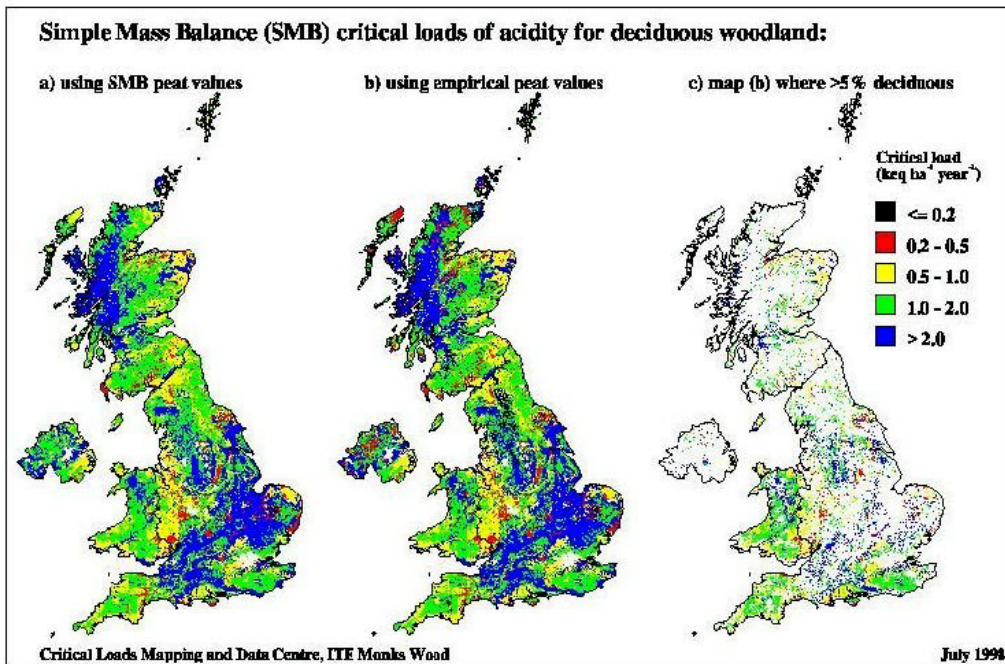
Figure 3



Data acknowledgement: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

Figure 4

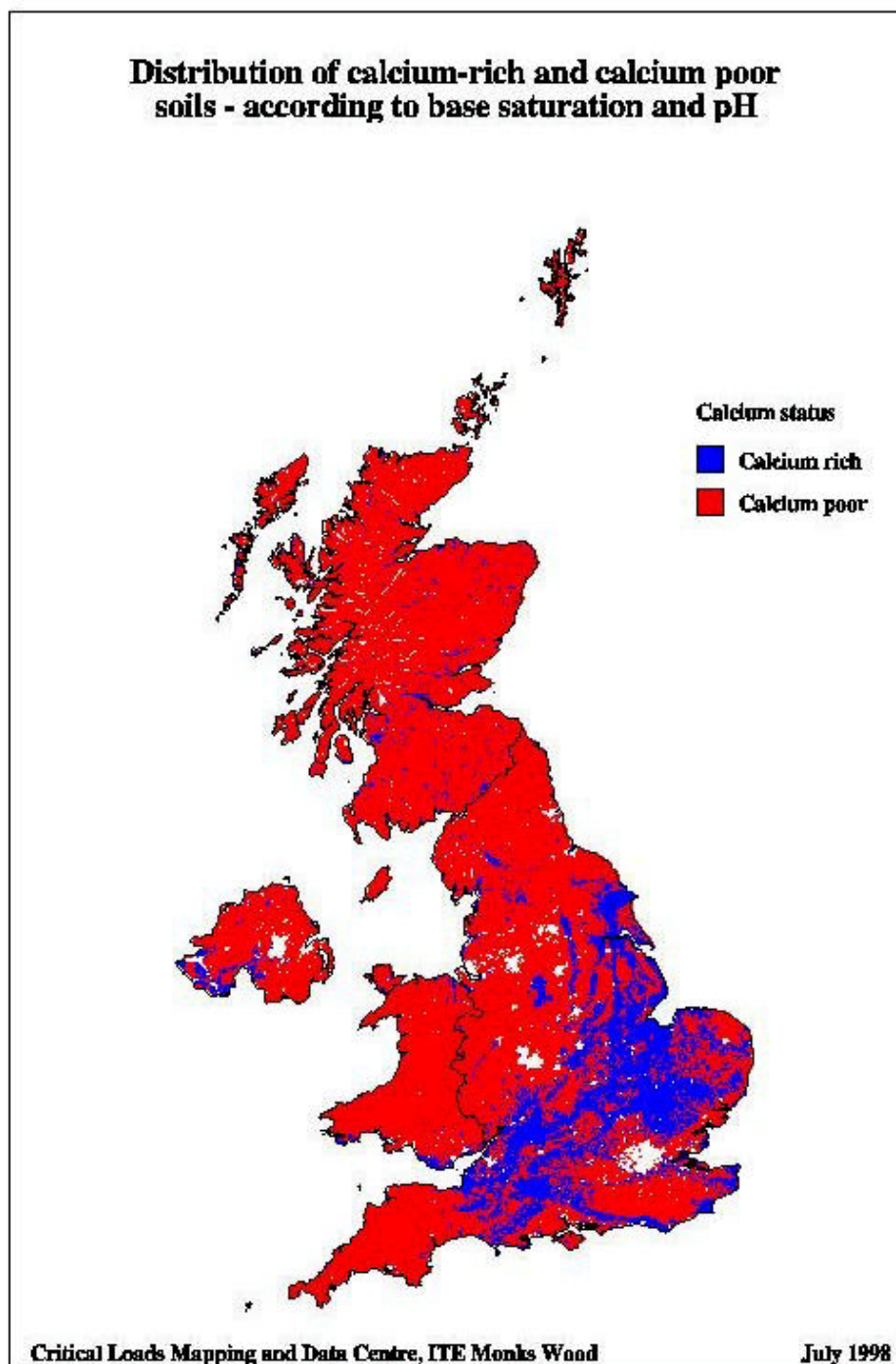


Data acknowledgement: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

Figure 5

Figure 6



Data acknowledgement: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

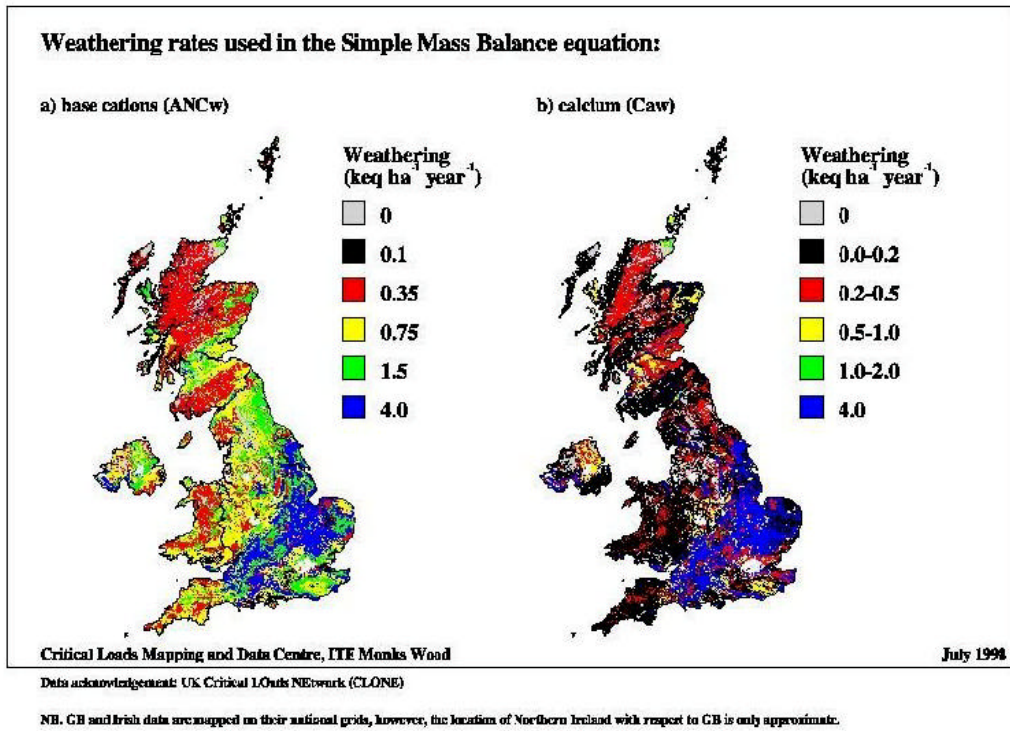


Figure 7

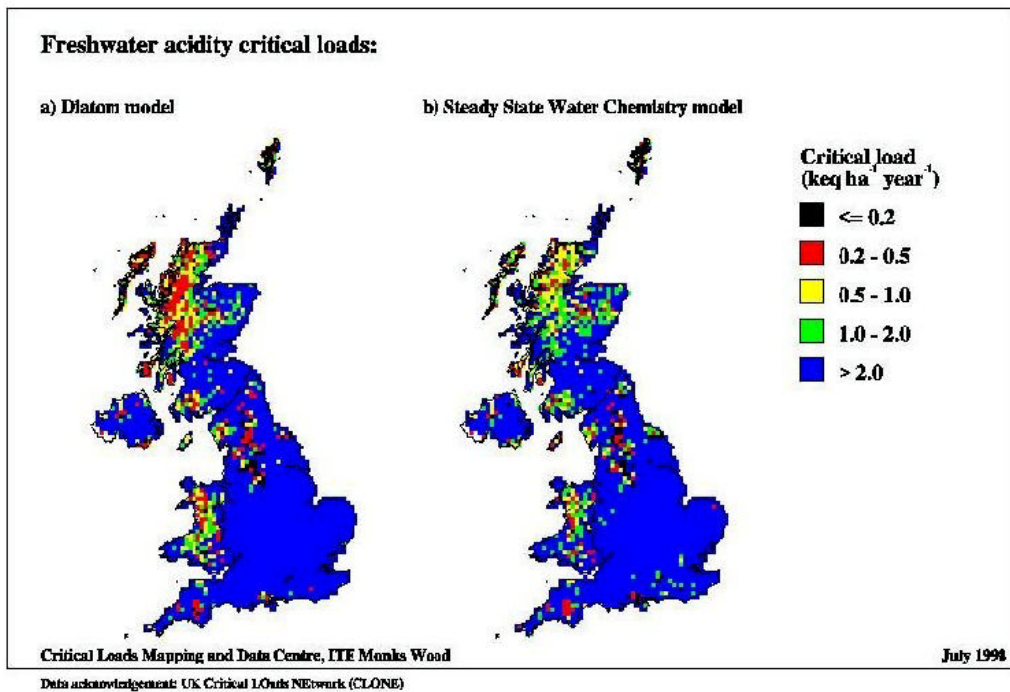


Figure 8

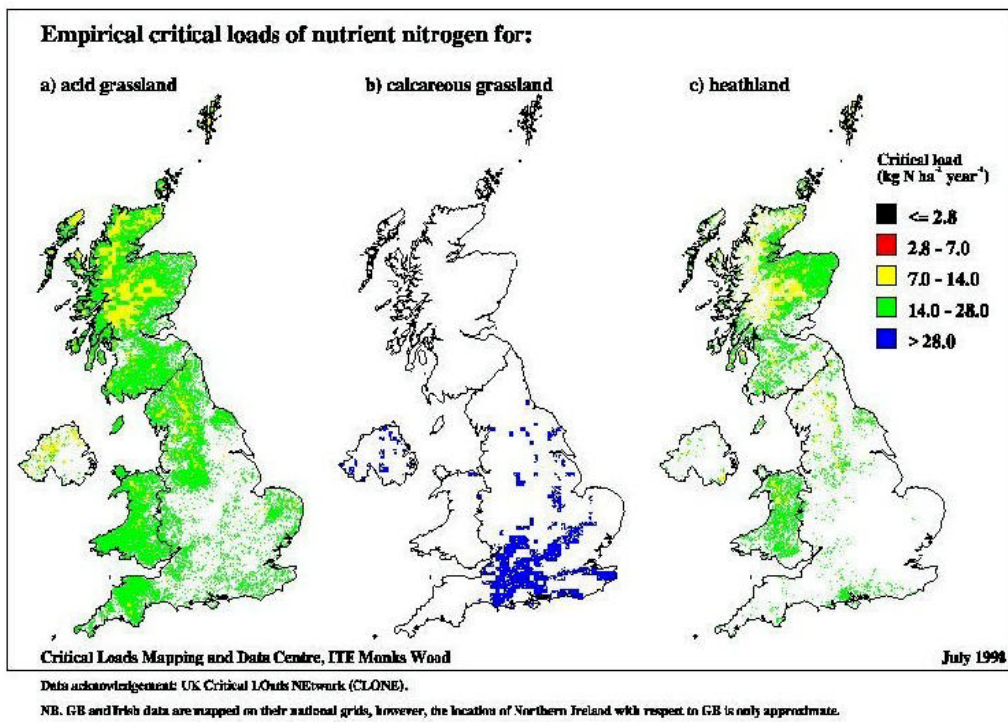


Figure 9

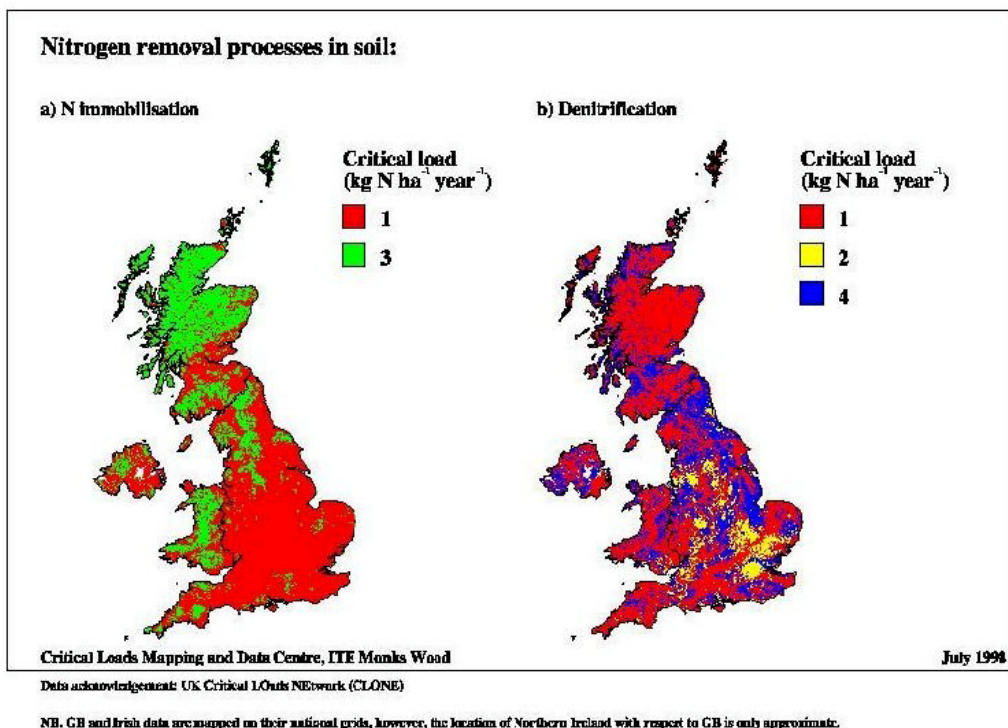
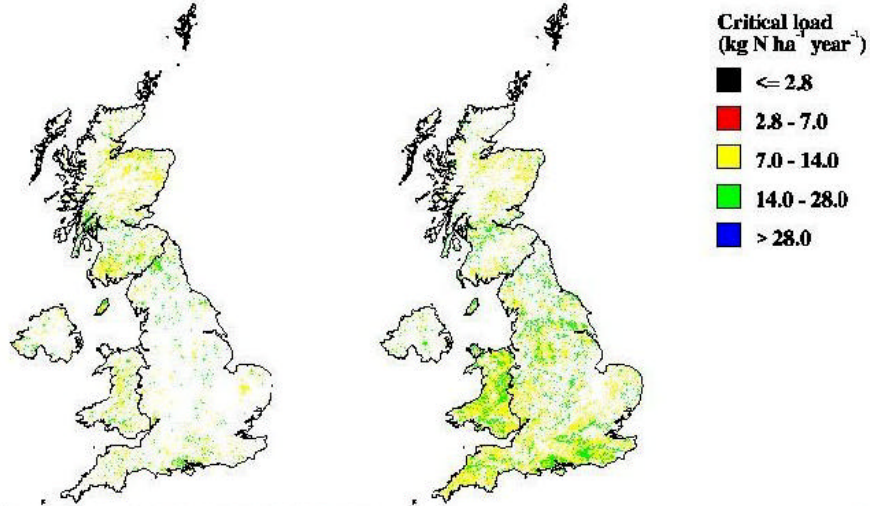


Figure 10

**Critical loads of nutrient nitrogen for woodland ecosystems:**

a) conifers (mass balance derived)

b) deciduous (combined empirical/mass balance)



Critical Loads Mapping and Data Centre, ITE Munks Wood

July 1998

Data acknowledgement: UK Critical Loads Network (CLONE)

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

Figure 11

The components of the Critical Loads Function

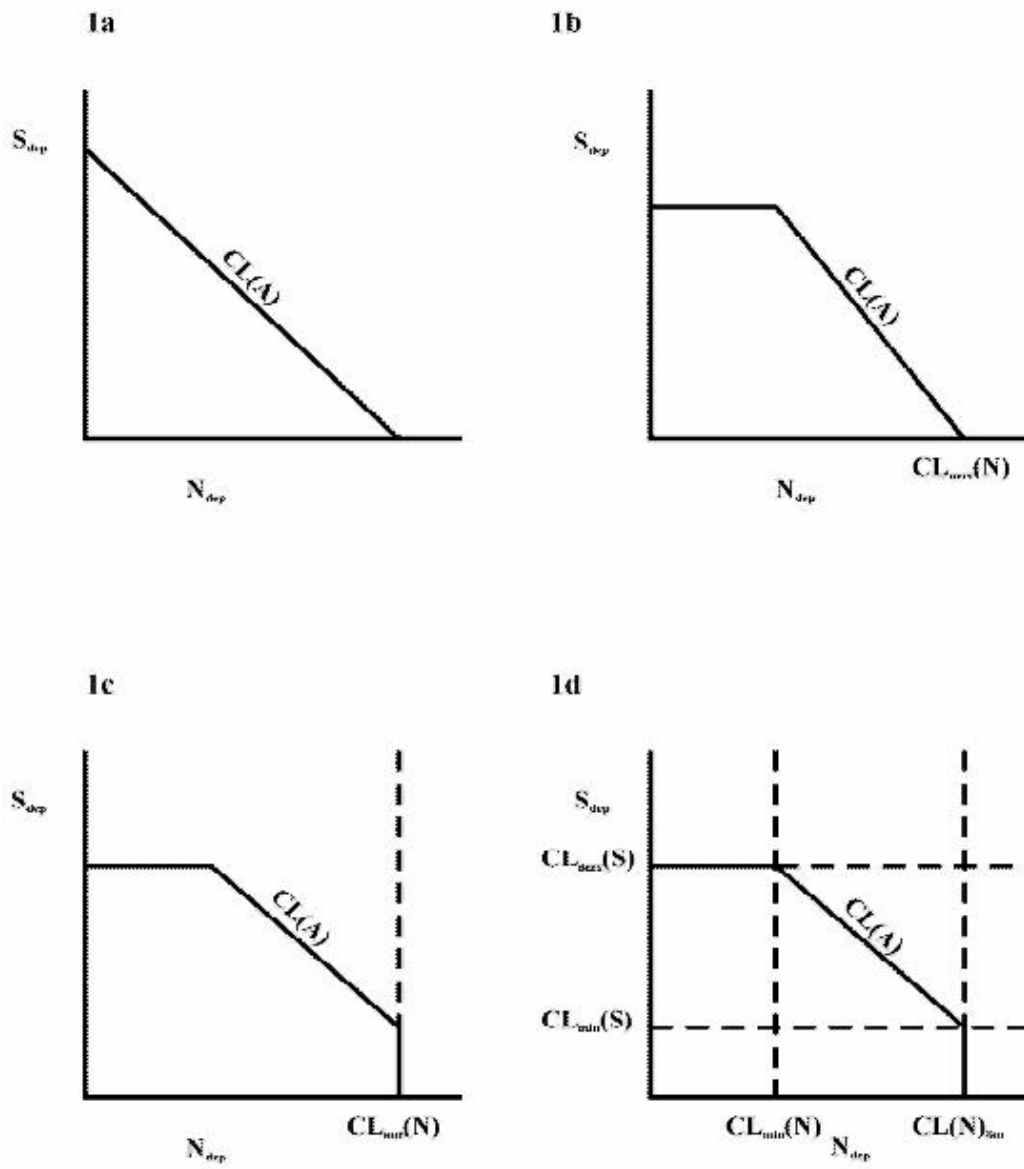
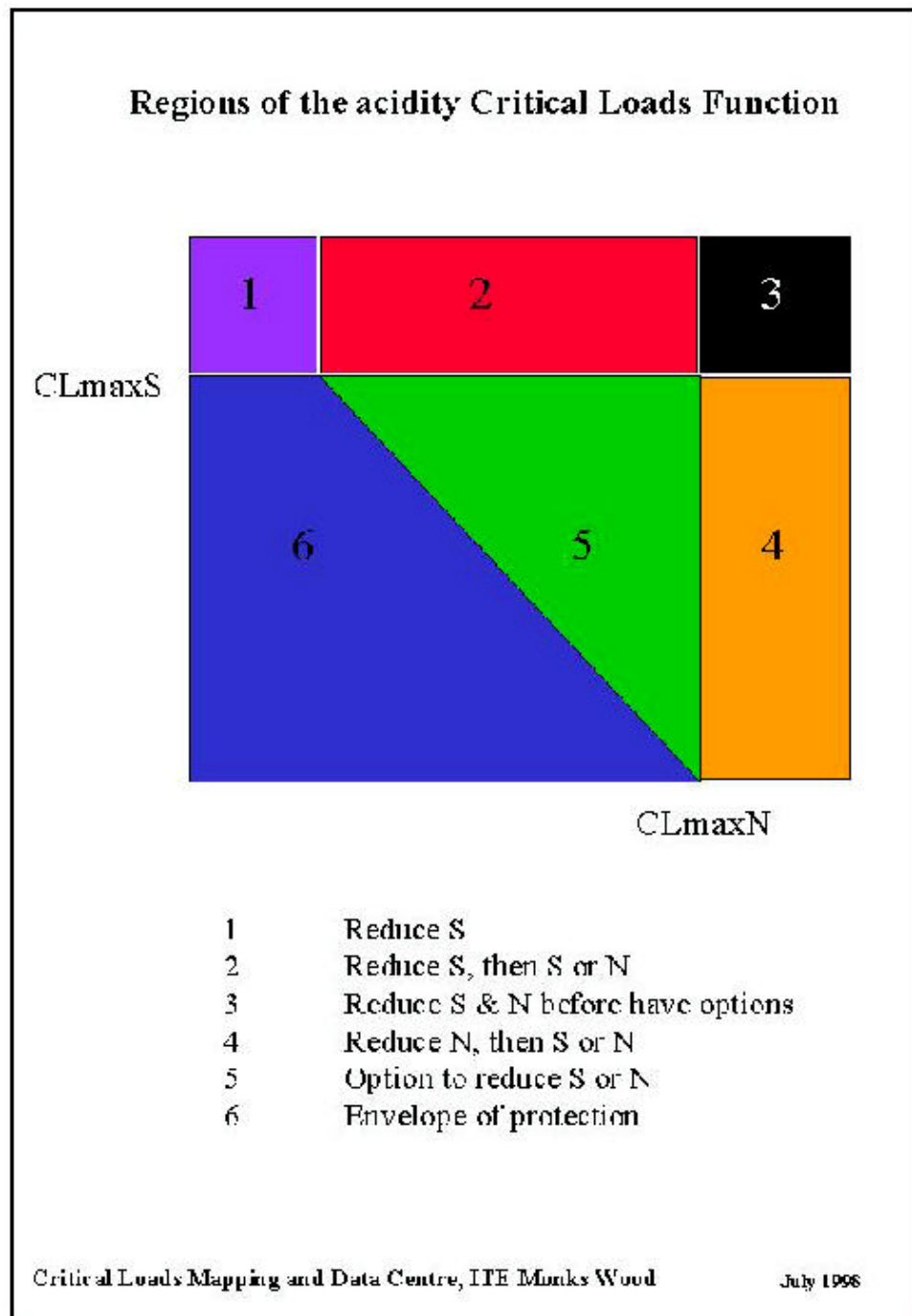
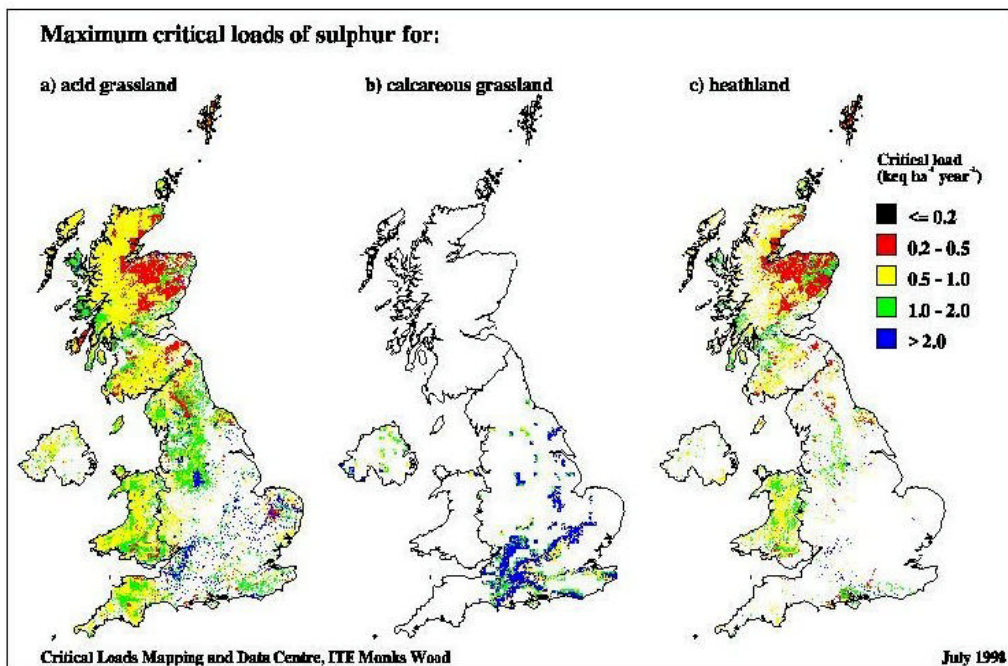




Figure 13

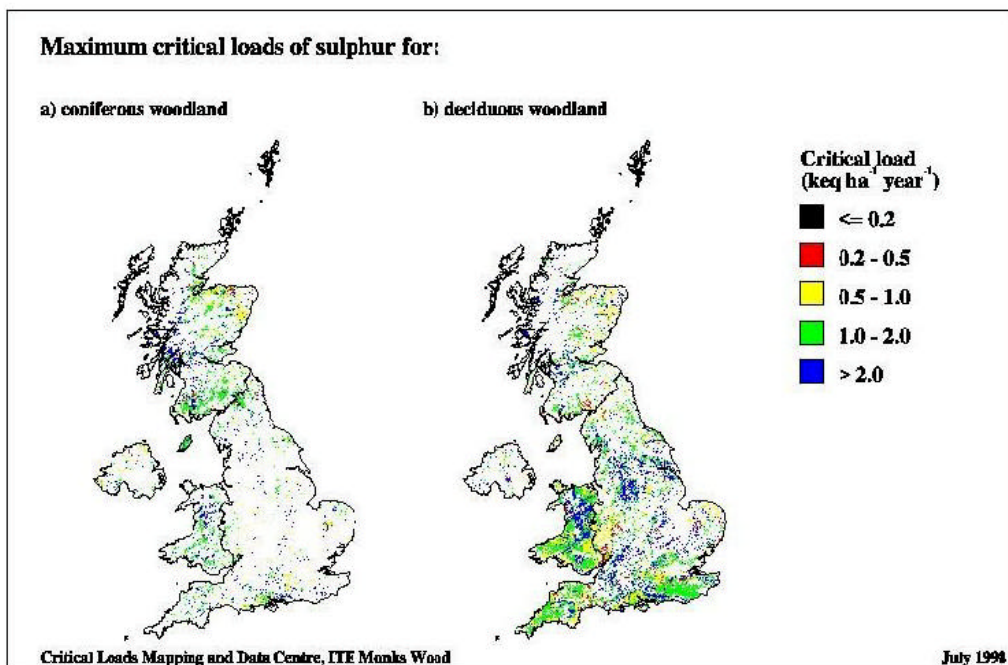




Data acknowledgement: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

Figure 14

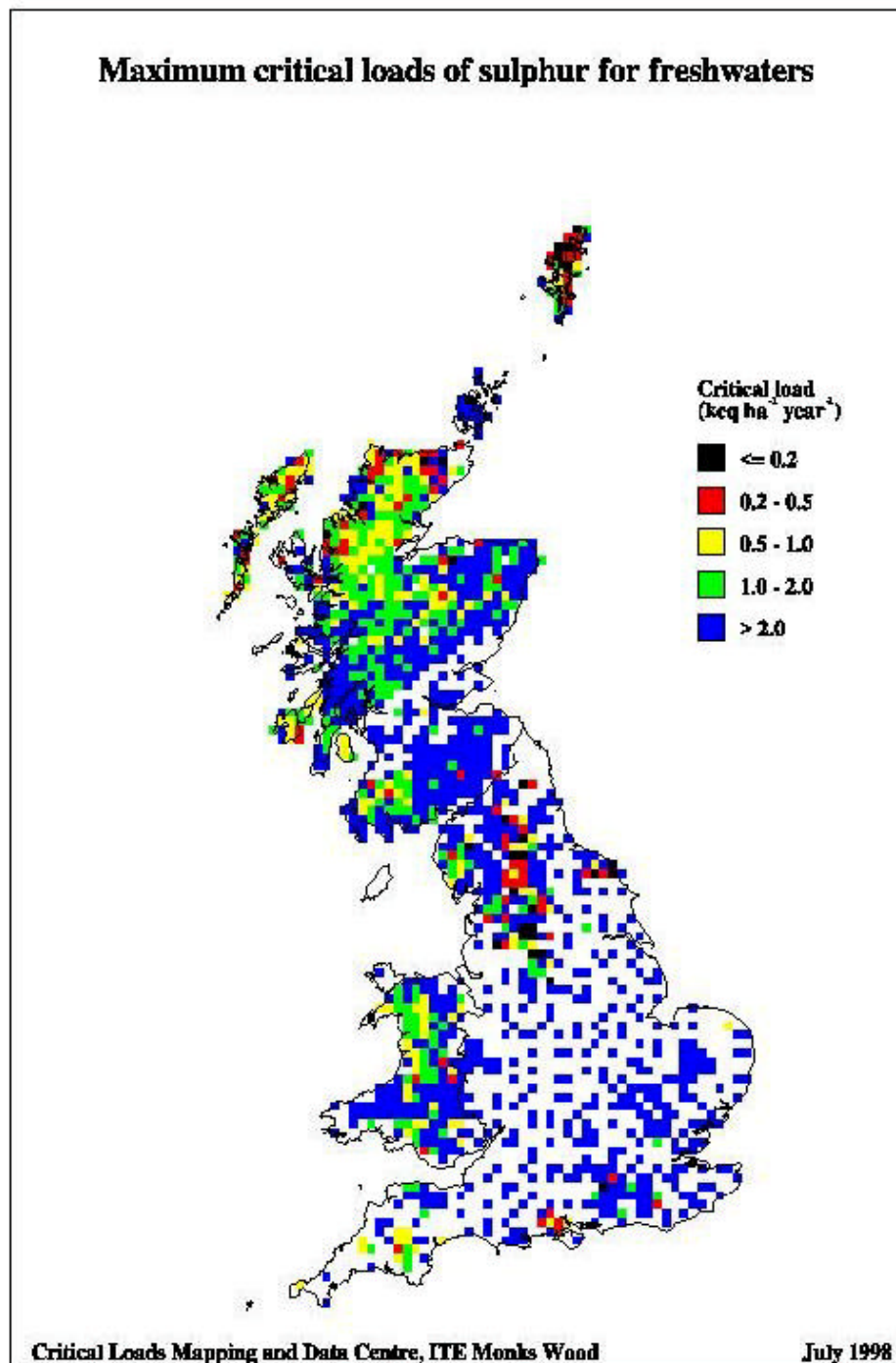


Data acknowledgement: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

Figure 15

Figure 16



Data acknowledgment: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

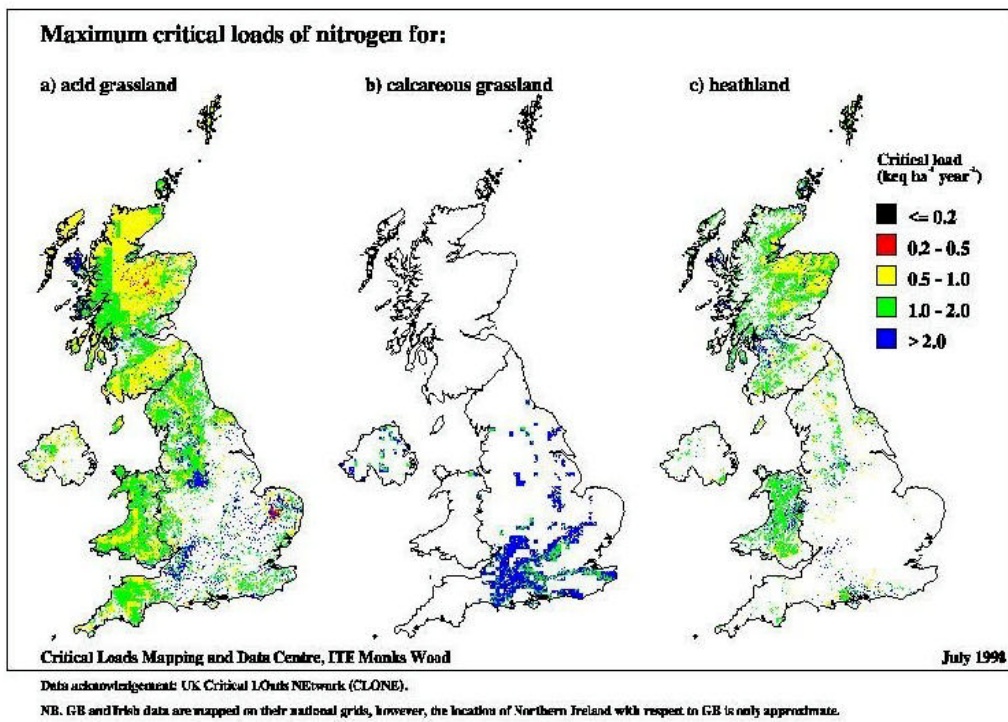


Figure 17

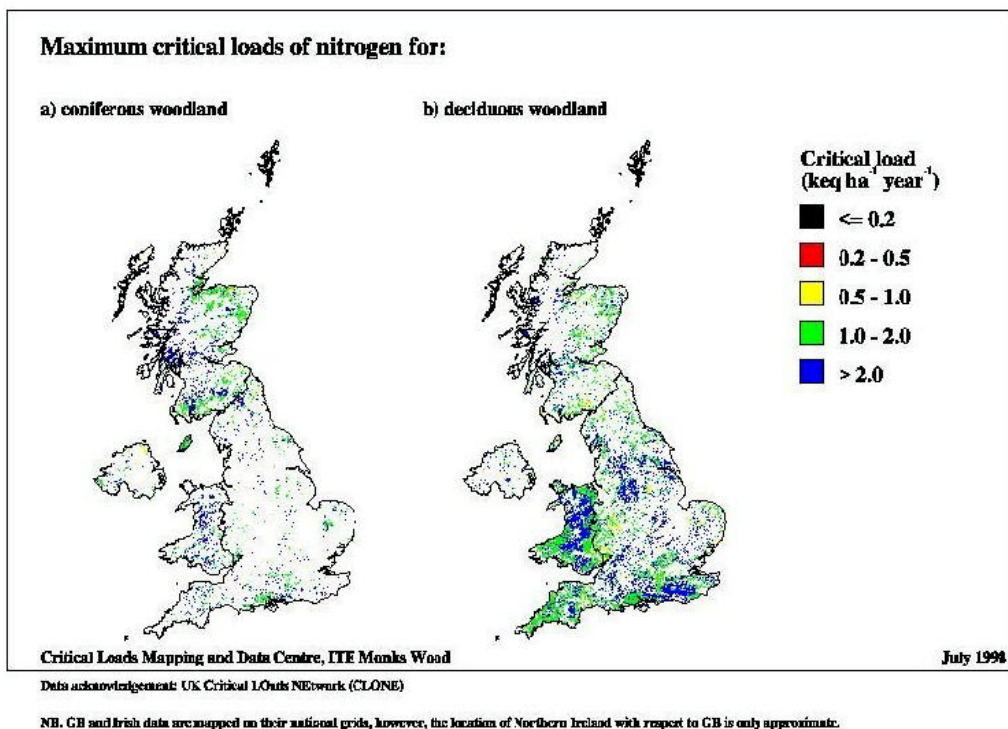
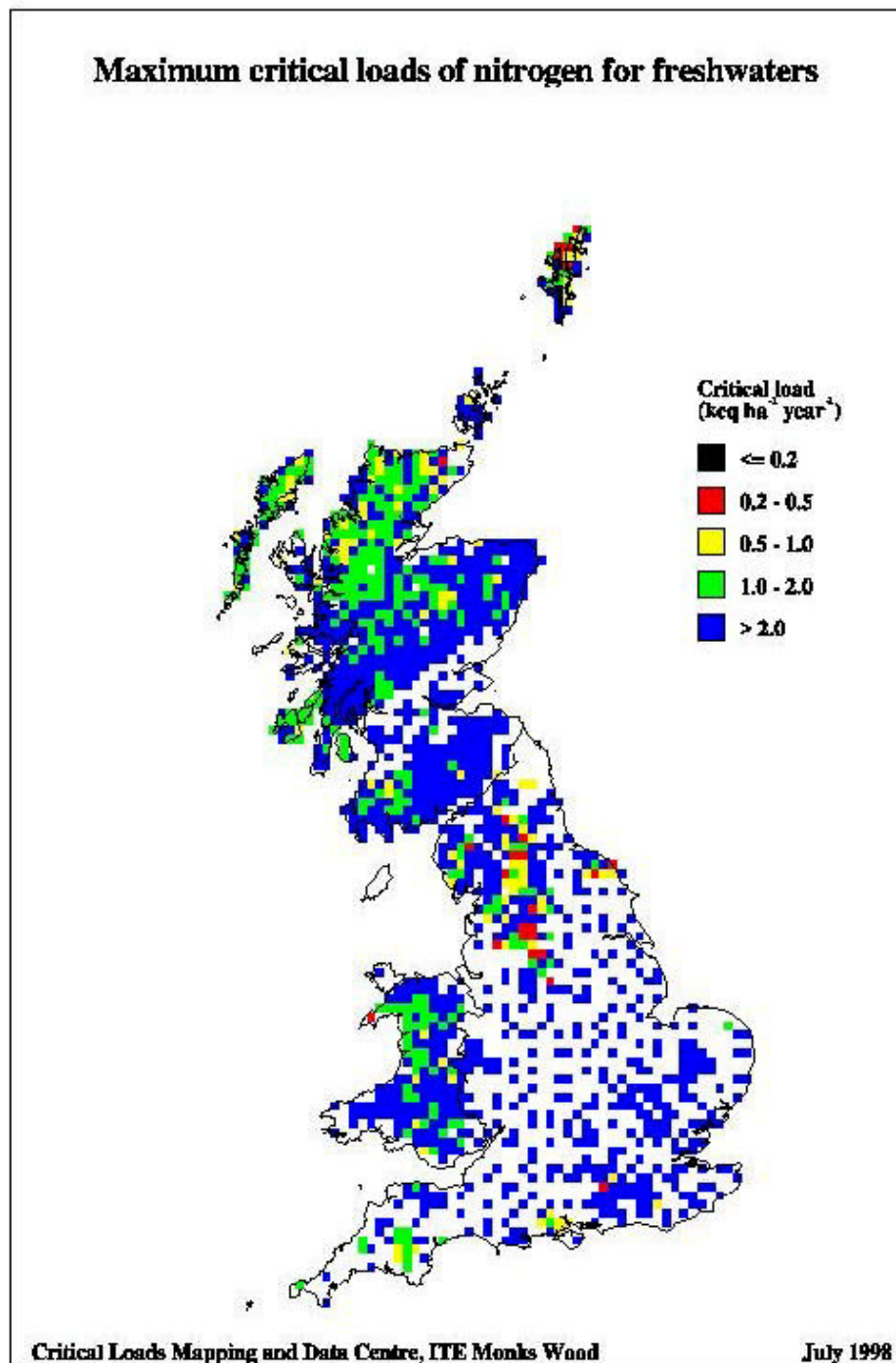


Figure 18

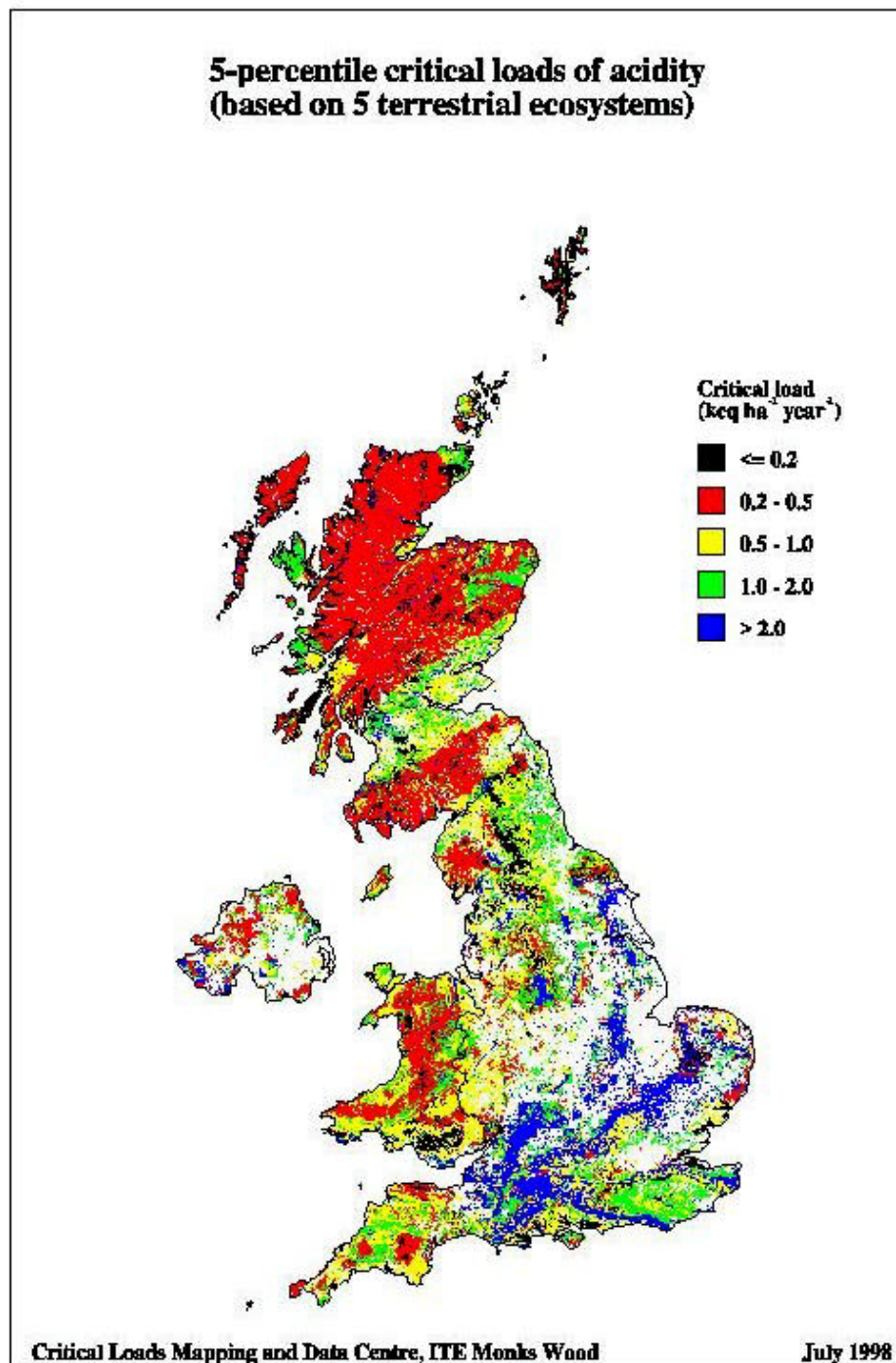
Figure 19



Data acknowledgment: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

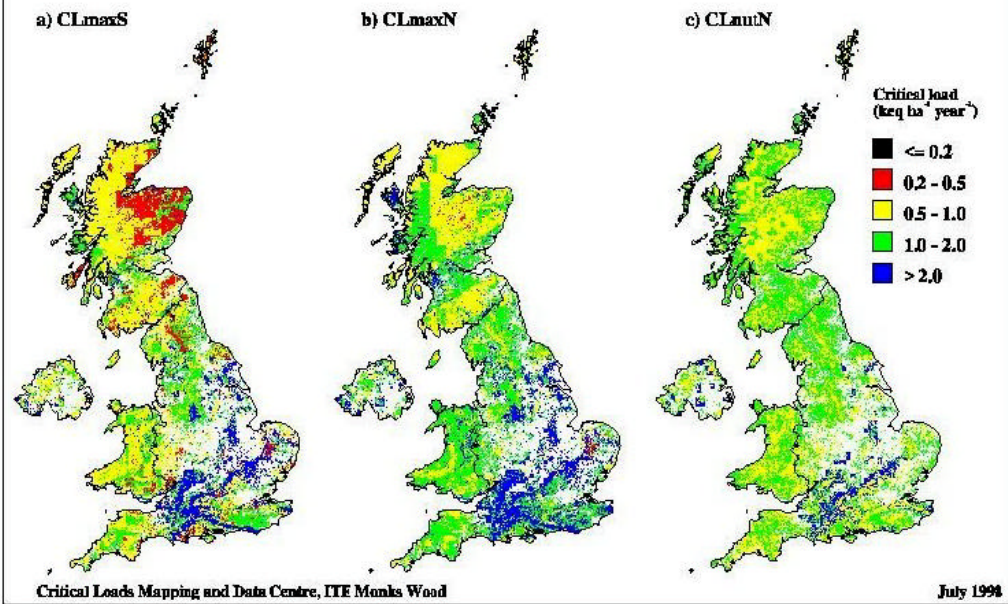
Figure 20



Data acknowledgment: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

5-percentile critical loads (based on 5 terrestrial ecosystems):



Data acknowledgment: UK Critical Loads Network (CLONE).

NB. GB and Irish data are mapped on their national grids, however, the location of Northern Ireland with respect to GB is only approximate.

Figure 21

## GLOSSARY

$ANC_{le(crit)}$	Critical leaching of base cations
$ANC_w$ (or $BC_w$ )	Weathering of base cations (Acid Neutralising Capacity produced by weathering of soils)
$BC:Al$	Base cation to aluminium ratio in soil solution
$BC_{dep}$	Base cation deposition (from the atmosphere)
$BC_u$	Base cation uptake (by plants)
$Ca:Al$	Calcium to aluminium ratio in soil solution
$Ca_{dep}$	Calcium deposition from the atmosphere
CCE	Co-ordination Centre for Effects
$CL(A)$	Critical load of acidity
CLF	Critical Loads Function (a function allowing critical loads to be considered in terms of sulphur and nitrogen deposition simultaneously)
$CL_{max}(N)$	Maximum critical load of nitrogen (a point on the Critical Loads Function equivalent to a critical load for acidity expressed in terms of nitrogen deposition and allowing for net base cation deposition)
$CL_{max}(S)$	Maximum critical load for sulphur (a point on the Critical Loads Function equivalent to a critical load for acidity expressed in terms of sulphur deposition and allowing for net base cation deposition)
$CL_{min}(N)$	Minimum critical load of nitrogen (a point on the Critical Loads Function equivalent to soil nitrogen removal processes, ie the acidity critical load before weathering is considered)
$CL(N)_{lim}$	Limiting critical load of nitrogen (a point on the combined acidity and nutrient nitrogen Critical Loads Function defined by a) the acidity in terms of nitrogen ( $CL_{max}(N)$ ) and b) the nutrient nitrogen critical load)
$CL_{min}(S)$	Minimum critical load of sulphur (a point on the Critical Loads Function defined by the intercept of the acidity and nutrient nitrogen Critical Load functions at the point where nutrient nitrogen deposition is equal to $CL(N)_{lim}$ )
$CL_{nut}(N)$	Critical load of nutrient nitrogen
CLRTAP	Convention on Long-Range Transboundary Air Pollution
EMEP	European Monitoring & Evaluation Programme (or more formally the Co-operative programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe)
FAB	First-order Acidity Balance Model



$L_{crit}$	Critical leaching flux of acid anions for the FAB model
MLURI	Macaulay Land Use Research Institute (Aberdeen)
$N_{de}$	Denitrification (from soils)
$N_{dep}$	Nitrogen deposition (from the atmosphere)
NFC	National Focal Centre
$N_i$	Nitrogen immobilisation (in soils)
$N_{le(acc)}$	Acceptable level of nitrogen leaching (from soils)
$N_u$	Nitrogen uptake (by plants)
$S_{dep}$	Sulphur deposition (from the atmosphere)
SMB	Simple Mass Balance (model)
SSLRC	Soil Survey and Land Research Centre (Silsoe)
SSWC	Steady State Water Chemistry (model)
TFIAM	Task Force on Integrated Assessment Modelling
TFM	Task Force on Mapping

## APPENDIX 1

SMB Equation using Ca:Al ratio as chemical criterion.

NB Base cation (BC) terms used here only relate to calcium.

$$CL(A) = ANC_w - ANC_{le(crit)}$$

where:

$CL(A)$  = critical loads of acidity (calculated in eq ha<sup>-1</sup> year<sup>-1</sup>) [using units given here divide by 1000 to give keq ha<sup>-1</sup> year<sup>-1</sup>]

$ANC_w$  = Acid Neutralising Capacity produced by weathering (eq ha<sup>-1</sup> year<sup>-1</sup>)

$ANC_{le(crit)}$  = critical leaching of ANC (eq ha<sup>-1</sup> year<sup>-1</sup>) =  $-Al_{le(crit)} - H_{le(crit)}$

$Al_{le(crit)}$  = critical leaching of Aluminium (eq ha<sup>-1</sup> year<sup>-1</sup>) =  $((1.5 * BC_{le}) / Ca:Al) * 1000$

$BC_{le}$  = calcium leaching (keq ha<sup>-1</sup> year<sup>-1</sup>) =  $BC_a - BC_u$

$BC_u$  = net uptake of calcium (keq ha<sup>-1</sup> year<sup>-1</sup>) = minimum( $u, BC_a$ )

$u$  = calcium uptake (keq ha<sup>-1</sup> year<sup>-1</sup>), see values in Table 1

$BC_a$  = calcium availability (keq ha<sup>-1</sup> year<sup>-1</sup>) = maximum( $Ca_w + Ca_{dep} - BC_{lemin}, 0$ )

$Ca_w$  = calcium weathering (keq ha<sup>-1</sup> year<sup>-1</sup>)

$Ca_{dep}$  = total (marine plus non-marine) calcium deposition for woodland 1992-94 (keq ha<sup>-1</sup> year<sup>-1</sup>)

$BC_{lemin}$  = minimum calcium leaching (keq ha<sup>-1</sup> year<sup>-1</sup>) =  $Q * [BC_l] * 0.01$

$Q$  = runoff (metres year<sup>-1</sup>)

$[BC_l]$  = limiting concentration of uptake of calcium (2 meq l<sup>-1</sup>)

$He_{le(crit)}$  = critical leaching of hydrogen ions (eq ha<sup>-1</sup> year<sup>-1</sup>) =  $(1.5 * ((BC_{le} * 1000) / (K_{gibb} * Ca:Al)))^{1/3} * (Q * 10000)^{2/3}$

$K_{gibb}$  = gibbsite equilibrium constant (950 [m<sup>6</sup>/eq<sup>2</sup>])

Ca:Al = Calcium:Aluminium ratio = 1

## APPENDIX II

Ranges of empirical critical loads of nutrient nitrogen agreed by the UNECE Task Force on Mapping in December 1995 and the mapping values used in the UK.

Ecosystem	Critical loads kg N ha <sup>-1</sup> year <sup>-1</sup>	
	TFM ranges	UK mapping value
Soft water lake	5-10 ##	not mapped
Mesotrophic fens	20-35 #	20
Ombrotrophic bogs	5-10 ##	10
Calcareous species-rich grassland - N or P limited	15-35 #	50
Neutral-acid species-rich grassland	20-30 #	25
Montane sub-alpine grassland	10-15 (#)	12
Lowland dry heath	15-20 ##	17
Lowland wet heath	17-22 #	17
Species-rich heath / acid grassland	10-15 #	17
Upland Calluna heaths	10-20 (#)	15
Arctic / alpine heaths	5-15 (#)	10
Acidic coniferous forest - nutrient imbalance	10-15 ##	mass balance value
- nitrogen saturation	10-25 ##	
- ground flora changes	7-20 ##	
Acidic deciduous forest - nutrient imbalance	15-20 #	17
- ground flora changes	10-20 (#)	
Deciduous calcareous forest	15-20 (#)	17

## reliable, # quite reliable, (#) expert judgement

**APPENDIX III**

## Classes of the ITE Land Cover Map

<b>Class</b>	<b>Description</b>
1	Sea
2	Inland water
3	Coastal bare ground
4	Saltmarsh
5	Lowland grass heath
6	Pasture/amenity grass
7	Meadow/verge/unimproved grass
8	Marsh/rough grass
9	Montane/hill grass
10	Dwarf shrub/grass moorland
11	Upland dwarf shrub moorland
12	Bracken
13	Lowland heath
14	Scrub/orchard
15	Deciduous wood
16	Coniferous wood
17	Upland bog
18	Arable/tilled land
19	Ruderal weeds
20	Suburban
21	Urban
22	Bare ground
23	Felled forest
24	Lowland peat bog
25	Dwarf shrub/grass heath

**APPENDIX IV**

Nitrogen immobilisation values for different soil types

<b>Soil code</b>	<b>Soil description</b>	<b>kg N/ha/year</b>	<b>keq N/ha/year</b>
1	Terrestrial raw soil	3	0.2143
2	Raw gley soils	1	0.0714
3	Lithomorphic soils	1	0.0714
3.1	Rankers	1	0.0714
3.2	Sand rankers	1	0.0714
3.4	Rendzinas	1	0.0714
3.6	Sand parendzinas	1	0.0714
3.7	Rendzina-like alluvial soils	1	0.0714
4.1	Calcareous pelosols	1	0.0714
4.2	Non-calcareous pelosols	1	0.0714
4.3	Argillic pelosols	1	0.0714
5.1	Brown calcareous earths	1	0.0714
5.2	Brown calcareous sands	1	0.0714
5.3	Brown calcareous alluvial soil	1	0.0714
5.4	Brown earths	1	0.0714
5.5	Brown sands	1	0.0714
5.6	Brown alluvial soils	1	0.0714
5.7	Argillic brown earths	1	0.0714
5.8	Paleo-argillic brown earths	1	0.0714
6.1	Brown podsollic soils	3	0.2143
6.3	Podsols	3	0.2143
6.4	Gley podsols	3	0.2143
6.5	Stagnopodsols	3	0.2143

7.1	Stagnogley soils	1	0.0714
7.2	Stagnohumic gley soils	3	0.2143
8.1	Alluvial gley soils	1	0.0714
8.2	Sandy gley soils	1	0.0714
8.3	Cambic gley soils	1	0.0714
8.4	Argillic gley soils	1	0.0714
8.5	Humic-alluvial gley soils	1	0.0714
8.6	Humic-sandy gley soils	1	0.0714
8.7	Humic gley soils	1	0.0714
9	Man-made soils	1	0.0714
9.2	Disturbed soils	1	0.0714
10.1	Raw peat soils	3	0.2143
10.2	Earthy peat soils	3	0.2143
2.2	Unripened gley soils	1	0.0714
1.1	Raw sands	1	0.0714
4	Pelosols	1	0.0714
5	Brown soils	1	0.0714
6	Podzolic soils	3	0.2143
7	Surface-water gley soils	1	0.0714
8	Ground-water gley soils	1	0.0714
10	Peat soils	3	0.2143

**APPENDIX V**

Denitrification values for different soil types

Soil code	Soil description	kg N/ha/year	keq N/ha/year
1	Terrestrial raw soil	1	0.0714
2	Raw gley soils	1	0.0714
3	Lithomorphic soils	1	0.0714
3.1	Rankers	1	0.0714
3.2	Sand rankers	1	0.0714
3.4	Rendzinas	1	0.0714
3.6	Sand parendzinas	1	0.0714
3.7	Rendzina-like alluvial soils	1	0.0714
4.1	Calcareous pelosols	2	0.1429
4.2	Non-calcareous pelosols	2	0.1429
4.3	Argillic pelosols	2	0.1429
5.1	Brown calcareous earths	1	0.0714
5.2	Brown calcareous sands	1	0.0714
5.3	Brown calcareous alluvial soil	1	0.0714
5.4	Brown earths	1	0.0714
5.5	Brown sands	1	0.0714
5.6	Brown alluvial soils	1	0.0714
5.7	Argillic brown earths	1	0.0714
5.8	Paleo-argillic brown earths	1	0.0714
6.1	Brown podsollic soils	1	0.0714
6.3	Podsols	1	0.0714
6.4	Gley podsols	1	0.0714
6.5	Stagnopodsols	1	0.0714
7.1	Stagnogley soils	4	0.2857
7.2	Stagnohumic gley soils	4	0.2857
8.1	Alluvial gley soils	4	0.2857
8.2	Sandy gley soils	4	0.2857
8.3	Cambic gley soils	4	0.2857
8.4	Argillic gley soils	4	0.2857
8.5	Humic-alluvial gley soils	4	0.2857
8.6	Humic-sandy gley soils	4	0.2857
8.7	Humic gley soils	4	0.2857
9	Man-made soils	1	0.0714
9.2	Disturbed soils	1	0.0714
10.1	Raw peat soils	1	0.0714
10.2	Earthy peat soils	1	0.0714
2.2	Unripened gley soils	4	0.2857
1.1	Raw sands	1	0.0714
4	Pelosols	2	0.1429
5	Brown soils	1	0.0714
6	Podzolic soils	1	0.0714
7	Surface-water gley soils	4	0.2857
8	Ground-water gley soils	4	0.2857
10	Peat soils	1	0.0714